# An assessment of rehabilitation gravels for *Salmo trutta* spawning: a case study from a small chalk stream, the River Stiffkey, Norfolk, UK.

Thesis submitted for the degree of

Doctor of Philosophy

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I, Luke Mitchell, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

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## Abstract

UK salmonid stocks have shown a sharp decline over the past 50 years. River channel modification, land-use intensification and change in agricultural practices are significant factors that have contributed towards this decline. An associated accumulation of fine grained sediment in spawning substrate inhibits population recruitment at the embryo stage of the life-cycle. Additions of rehabilitation gravel to the River Stiffkey, a small chalk stream in North Norfolk, in 2003 by the Wild Trout Trust (WTT) and again in 2009 as part of the Living North Sea project aimed to augment migratory *Salmo trutta* L. (sea trout) populations. Rehabilitation by means of gravel introduction has anecdotal short-term benefits but physical environmental constraints at various spatial scales over the medium- to long-term have not yet been adequately quantified. In order to better understand the role of rehabilitation gravel in the reproduction and recruitment of *S. trutta* populations, this study examined: the physical suitability and morphosedimentary nature of rehabilitation gravel as a spawning habitat, embryo survival within rehabilitation gravel and sedimentary constraints that limit recruitment at this early life-stage, catchment controls that define the physical character of the river, and the spatial relationship between key juvenile life-stage dependent habitat types. River Stiffkey rehabilitation gravel was installed to similar specifications in 2003 and in 2009 and as such provided a spatial and temporal assessment of physical and biological variability. Results indicate the importance of catchment controls and historic regulation in determining channel processes. Rehabilitation gravel underwent a sediment composition succession from an unstable well sorted gravel (40-10 mm) type to a poorly sorted stable deposit as fine sediment (<1 mm) was deposited and surface gravels eroded. Embryo survival declined as fine sediment (<1 mm) accrued and permeability decreased. Rehabilitation gravel was characterised by a net loss of small sized gravel ($30>D_{50}\geq16$ mm) required for spawning by non-migratory *S. trutta* populations and accrued an abundance of fine grained sediments ($D<1$ mm) over time. Rehabilitation gravel installed in 2003 had consistently poor embryo survival, whilst gravel installed in 2009 had a positive response to a reduced sediment load stress. Consequently, rehabilitation gravel in lowland chalk stream catchments characterised by high diffuse inputs of agricultural sediment may have a short (<10 years) lifespan. As such rehabilitation gravel is likely to have a limited role in *S. trutta* recruitment. Recruitment of *S. trutta* in the River Stiffkey was regulated by both poor abundances of key life-stage dependent habitat, and spatial connectivity between them. A river rehabilitation management approach based on a hierarchy of spatial scales that identifies and addresses ecological constraints to recovery in a systematic top-down approach from the catchment level to the macrohabitat is proposed.

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For Ava

time to play, cuddle and dream

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# 1 Introduction

## 1.1 Rivers: an introduction

Rivers are a unique and precious natural resource. Of the world's water resources <3% is freshwater, the majority of which is stored in the polar ice caps (70%) and groundwater reserves (30%) leaving a mere 0.0001% of the total global water budget as river water (Giller and Malmqvist, 1998). Nevertheless, fluvial systems remain key surface processes eroding, transporting and depositing approximately 20 billion tons of sediment each year (Leopold et al., 1964; Press and Siever, 1998; Easterbrook, 1999). Streams and rivers are particularly heterogeneous systems ranging in temporal and spatial scale from the microhabitat (cm) to the entire catchment (km) (Frissell et al., 1986, Kondolf and Larson, 1995). Human development has a history of dependence on rivers that continues today and as such conflicts with environmental needs of this scarce resource. River systems and their associated floodplains provide water for hydropower, the manufacturing industry, agriculture, domestic consumption, navigation ways, amenity, human settlement and a conduit for effluent. As such rivers possess great economic (see Clark, 2005), socio-cultural (see Pooley, 2005) and political value. In order to exploit riverine resources and to control for flood and drought risk, human impacts on channel morphology have, through the centuries, been great (Boon, 1992). Such modification was not sympathetic of the aquatic ecosystem and has had deleterious impacts on the physical, biological and chemical components resulting in low diversity and an unproductive ecological system.

Advances in our understanding of fluvial processes have led to a paradigm change in our approach to river management with greater emphasis on a holistic socio-ecological approach. Restoration of ecosystem structure and function is currently a statutory requirement in the European Union whilst the challenge of water resource use and flood mitigation gathers pace in our changing environment. Policy makers and river managers are increasingly looking towards the principles of hydrogeomorphology to bridge the divide and provide integrated and sustainable solutions to river management going into the 21st century.

### 1.1.1 Rivers of Great Britain

Great Britain has many rivers due to its temperate climate and geology (Gregory, 1997; Park, 2005b). Rivers are typically short in global terms (Park, 2005a) and have annual low flows punctuated by short duration flood flows (Folkard, 2005a). British rivers are characterised by climatic, geological, and topographical differences. Upland rivers typical of Scotland and Wales have steeper gradients, are rain fed and drain an impermeable geology, while the south and east lowland streams of England have a gentle gradient, are aquifer fed and drain a permeable, often chalk geology. North and western areas receive greater precipitation due to characteristic topographic features and, with a larger proportion of the population living in the south and east, rivers in these areas are subject to greater anthropogenically derived environmental pressures and constraints such as abstraction and channel modification.

British rivers have a long history of anthropogenic modification and just about all have been impacted either directly or indirectly through channel modification, impoundments, inter-basin transfers and/or abstractions (Higgs and Petts, 1988; Petts, 1988). Deforestation of British river catchments between 3000-2000 BC (Sheail, 1988; Gregory and Davis, 1997) altered catchment process dynamics. However, it was not until the 1st century AD under Roman occupation that direct large scale river modification began with the land drainage and flood embankments schemes of the Somerset Levels and East Anglia Fens (Park, 2005a; Watson, 2005). The Doomsday survey of 1086 indicated that rivers were increasingly used to harness power with >5000 water mills recorded (Sheail, 1988). From the 12th century, river channels were being modified for navigation, land drainage and water meadow development (Brookes et al., 1983; Eaton, 1989; Gregory and Davis, 1997; Clark, 2005). Further widespread regulation and land drainage in the East Anglian Fens took place during the 17th century (Sheail, 1988).

Britain was dominated by an agricultural economy towards the mid 18th century and had very low population pressures acting on this resource. However, industrialisation and urbanisation intensified pressures on river waters. River regulation began in the late 18th century with the impoundment of upland river catchments to maintain consistent water levels for hydropower (Higgs and Petts, 1988) and to facilitate industrial water demands (Mann, 1988). Large inter-basin schemes to transport water to urbanised regions of the south east were developed between the 18th and 19th century (Eaton, 1989). As rivers became more industrialised towards the mid 19th century water quality rapidly deteriorated due to increasing levels of sewerage and trade effluent (Sheail, 1988). Between 1930-1980 the Royal Commission on

Land Drainage in England and Wales (1927) and the Land Drainage Act of 1930 provided the basis for major river channelisation. During this time up to 25% of main river channels were subjected to major engineering works (Brookes et al., 1983). To date, approximately 98% of all rivers in England and Wales have been modified (Figure 1.1) (Eaton, 1989; Brookes, 1995; Wade et al., 1998; Park, 2005a) and there are now very few rivers left in a completely natural state (Gregory, 1997; Gregory and Davis, 1997).


**Figure 1.1** Extent of river channelisation in England and Wales between 1930-1980 (Modified from Brookes et al., 1983).

### 1.1.2 Chalk streams

The Chalk of southern and eastern England is composed of organic calcite derived from the coccoliths of marine algae deposits laid down some 65 to 100 million years ago during the Cretaceous Period (Ekdale and Bromley, 1984; Berrie, 1992; Bond, 2012a). The composition of Cretaceous Chalk is uniform pure calcite of variable texture, hardness and thickness. The occurrence of flint within Cretaceous Chalk is a characteristic feature (Aldiss et al., 2012). Because the Chalk is porous (40% by volume), it is the most important aquifer of southern England with water residence times of >20 years (Berrie, 1992; Aldiss et al., 2012).

Chalk streams were formed during the deglaciation of the last Ice Age (Bond, 2012a). Initially chalk streams were anastomosed due to high levels of glacial debris and sediment. The successive reduction of melt-water subsequently lowered stream power, and associated geomorphic activity, forming single-thread channels. Chalk streams are now associated with limited stream flow, simple but stable drainage networks and little lateral and longitudinal landscape connectivity (Sear et al., 1999; Bond, 2012a), largely due to channel modification. Chalk streams characteristically rise from the Cretaceous Chalk aquifer of the southern and eastern regions of England (Figure 1.2). England represents most (c. 85%) of the chalk river resource on a global scale (The Wildlife Trusts, n.d.). Chalk stream flow is maintained by groundwater springs (>75%) with little contribution from surface run-off creating low order streams of short lengths relative to catchment areas (Bond, 2012b; Berrie, 1992). High width to depth ratios, long inter-riffle spacing and a naturally low suspended sediment load characterise chalk streams (Berrie, 1992; Sear et al., 1999). However, current land-use has significantly increased the fine sediment load to streams, frequently degrading stream ecosystem function. Using a source fingerprinting approach in two agricultural catchments, the Rosemaund and Smisby catchments, Russell et al. (2001) observed that in-channel sources contributed little to suspended sediment loads (ca. 10% sediment yield) with land surface sources contributing up to 65% suspended sediment yield. Currently, 77% of chalk streams in the UK do not meet the ecological status requirements as set out by the Water Framework (WWF, 2014).

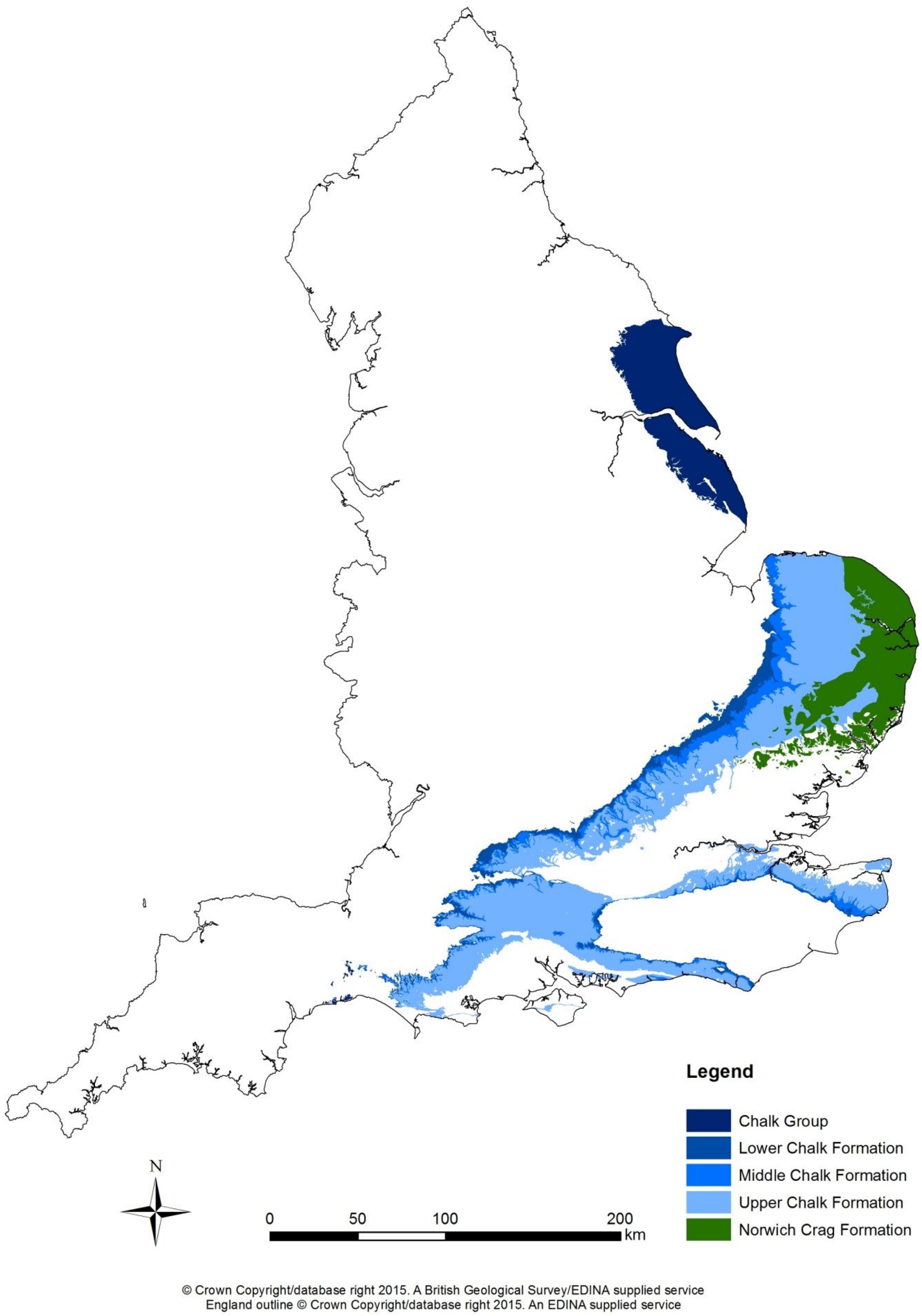



**Figure 1.2** Extent of the Cretaceous Chalk geology in England illustrating its southern and eastern extent. Glacial deposits and the Norwich Crag Formation, in the east of the Anglian region, are responsible for an increased flood hydrograph response in the chalk streams of North Norfolk.

The properties of the permeable Chalk geology give rise to distinctive physical and biological characteristics of chalk streams. Channel base flow is regulated through aquifer discharge forming characteristically hydrologically stable ecosystems. Aquifer recharge is maintained by winter precipitation. This increases groundwater levels and provides seasonal spring discharge higher up in the catchment leading to ephemeral upper catchment streams, commonly known as winterbournes, which flow only during winter (Berrie, 1992; Sear et al., 1999). During summer months groundwater levels typically decrease to base flow levels. Permeable chalk geology regulates the storm hydrograph reducing high flood response discharges common in surface fed streams (Sear et al., 1999). Peak flows of the typically moderated chalk stream hydrograph are lower than surface-fed stream types. However, these flows have a relatively extended duration due to elevated aquifer water levels following rainfall (Mainstone et al., 1999). Chalk rivers also maintain a stable and predictable thermal regime, between 5-17° C (Crisp et al., 1982; Mackey and Berrie, 1991). Chemical properties of the Chalk aquifer combined with a stable discharge sustain a constant chemical composition with a pH of 7.4-8.0 that maintain an ideal environment for primary production (Berrie, 1992). With limited stream power chalk streams have reduced geomorphic capacity for channel modification or coarse substrate transport resulting in few in-channel sediment storage bedforms (Sear et al., 1999). Chalk streams therefore typically have a surface armour layer of coarse grained sediments underlain by finer sediment (Frostick et al., 1984; Reid et al., 1997). Due to greater stream power, upland type streams have coarser grained substrate than is observed in the gravel deposits typically associated with chalk streams (Milan et al., 2000).

The characteristic chalk stream habitat is unique and has been the focus of many studies from a diversity of disciplines (Mann and Orr, 1969; Ladle and Bird, 1984; Marshall and Westlake, 1990; Berrie, 1992; Clarke et al., 2006). The high levels of ions released from the permeable underlying calcareous geology and clear water conditions supports an ecologically diverse range of flora and fauna (Mann et al., 1989; Berrie, 1992; Woodward et al., 2008). Typical chalk stream aquatic plants include *Callitriche* spp., *Zannichellia palustris* and *Ranunculus penicillatus* (Giller and Malmqvist, 1998; Wright and Symes, 1999). The European otter (*Lutra lutra*), water vole (*Arvicola terrestris*), brook lamprey (*Lampeta planeri*), bullhead (*Cottus gobio*) and white-clawed crayfish (*Austropotamobius pallipes*) are rare and designated UK Biodiversity Action Plan (BAP) priority species that are widely associated with chalk stream habitats (UK Steering Group, 1995b; Environment Agency, 2005). The invertebrate community of chalk streams is typically diverse with a very high biomass due to an abundant habitat and food resource within a stable environment (Wright, 1992). Additionally, chalk streams produce a high biomass of native *S. trutta* (Mann et al., 1989). Chalk streams are widely considered by UK conservation legislation as ecologically important and have been declared key habitats under the UK BAP (UK Steering Group, 1995a; Neal and Jarvie, 2005) and EU Habitats Directive to meet the ecological objectives and targets set out by the Water Framework Directive. The chalk streams of the UK are world renowned fisheries and have significant economic value. Salmonid angling in the UK is worth approximately £400 million (Clark, 2005). Over abstraction from Chalk aquifers has resulted in reduced low summer flows and, coupled with increased catchment derived sediment inputs, in-channel sediment deposition (Bond, 2012d). Over exploitation and anthropogenic degradation of valuable spawning habitats have had significant impact on *S. trutta* stocks (Hendry et al., 2003; Potter et al., 2003).

### 1.1.3 Chalk streams of North Norfolk

North Norfolk is underlain by an east dipping Cretaceous Chalk geology overlaid in the east by unconsolidated clays, silts, sands and gravels of the Pleistocene Crag aquifer (Figure 1.2) (Hiscock et al., 1996; Holman et al., 1999; Ander et al., 2006). The Chalk of North Norfolk, characterised by a mean porosity of 38%, is a very pure fine grained calcite with limited marl, clay and sand deposits (Price et al., 1976; Ander et al., 2006). Norfolk chalk is classed as a highly productive aquifer (Environment Agency, 2013). As the most important groundwater resource in East Anglia it is under considerable abstraction pressure (Ander et al., 2006).

The Rivers Burn, Stiffkey and Glaven are identified as the main regional river systems that drain the Chalk ridge spanning from Syderstone in the west to Cromer in the east. The Burn drains an area of 82 km$^2$ along its 13 km length, the River Stiffkey 140 km$^2$ through 30 km of stream length and the River Glaven drains an area of 115 km$^2$ through 17 km of stream channel. The Rivers Hun and Mundesley are smaller east flowing streams. Chalks streams typically have a low drainage density (Berrie, 1992) due to the high permeability index of the underlying chalk aquifer (Mainstone et al., 1999). They are short in relation to their catchment area size (Berrie, 1992).

Hydrogeomorphic controls on North Norfolk rivers have been modified through historical channel regulation and dredging. Their floodplains have been drained and their flows have been widely regulated by watermill and weir pool structures since the Roman era (Pawson, 2008). River morphology has thus been significantly modified with consequent bedform loss which, coupled with excessive inputs of fine grained sediment, has had deleterious ecological impacts.

Land use in the region is largely agricultural with wheat, potatoes, barley and sugar beet the dominant crops (Environment Agency, 2009; Ander et al., 2006). Key ecological concerns to river channels in the region are excessive catchment-derived fine (<1 mm) sediments (silts and clays) and associated poor land management. Roads act as conduits of agriculturally-derived sediment and have been identified as a major diffuse source of input (Natural England, 2013). Excessive deposits of sediment have had an adverse affect on the water quality of Rivers Burn, Stiffkey and Glaven (Environment Agency, 2009). Significant inputs of fine sediment in North Norfolk chalk streams are the result of considerable soil erosion in agricultural fields. These run-off events are largely triggered by convective rain storm events. North Norfolk chalk streams have elevated levels of fine sediment <1 mm (>25%), high quantities of medium sand ($D$ = 0.125-1 mm), and variable clay ($D$<0.063 mm) content (4.9-7.4%) (Milan et al., 2000). Spawning substrate has very high fine sediment yields by weight (>30%), with subsurface sediment typically containing as much as twice as that at surface levels (Milan et al., 2000).

Conservation bodies and charitable organisations formed in the last 15 years have provided the drive for river rehabilitation work in North Norfolk. The River Glaven Conservation Group (RGCG), formed in 1999, has been instrumental in delivering rehabilitating schemes on that river. In 2007 Fishery managers, scientists, landowners and local interest groups formed the Anglian Sea Trout Project (ARSTP) in response to declining habitat and diversity observed in Anglian rivers. As a collaborative and integrated partnership the ARSTP includes organisations such as the Wild Trout Trust (WTT), Environment Agency (EA), Centre for Environment, Fisheries, and Aquaculture Science (Cefas), the RGCG and the Norfolk Rivers Trust (NRT). Using migratory *S. trutta* as an indicator species the ARSTP aims to make habitat improvements on the three main North Norfolk Rivers: the Glaven, Stiffkey and Burn. The NRT, established in 2011, is a charitable organisation concerned with the rehabilitation and conservation of river ecosystems throughout Norfolk.

## 1.2 Principles of hydrogeomorphology

Fluvial geomorphology, or hydrogeomorphology, is the earth science concerned with river channel morphology and its response to physical processes at various spatial and temporal scales (Charlton, 2008). Rivers are dynamic open systems governed by energy and material exchange with the external environment. Climate, geology and catchment topography provide the energy that drives water, sediment and biological material exchange throughout the system (Knighton, 1984; Wharton, 2000). The dynamic and variable nature of rivers is attributable to system controls (Knighton, 1984). Any given fluvial system is regulated by variables operating from both within (internal controls) and outside (external controls) of the system. External variables such as climate, tectonic forces, base level (limit of channel downward erosion, frequently sea-level) and anthropogenic activities act independently of internal controls (Werritty, 1997). Conversely, internal controls (geology, soil, vegetation, catchment morphology, discharge) are influenced by external variables as well as other internal variables (Leopold et al., 1964; Charlton, 2008). In this manner the fluvial system exists in some dynamic quasi-equilibrium state. In a stable state control variables exist in relative equilibrium developed over time in response to the external environment (Wolman and Gerson, 1978; Hey, 1997). Alteration or disturbance of control variables, however, generate system disequilibrium and a geomorphic adjustment or a system response to an alternative equilibrium based on the altered control variables (Werritty, 1997). Such geomorphic adjustment is known as the process-response mechanism (Charlton, 2008).

River networks are hierarchically organised from larger scale stream systems through to the microhabitat system (Figure 1.3). System morphology at each level develops within a spatially and temporally nested hierarchy of scales that persists within the drainage catchment (Frissell et al., 1986). Larger scale river channel geometry persists over longer time scales and is less susceptible to change, while smaller scale channel forms respond readily and quickly to controlling processes (Frissell et al., 1986; Newson, 1992). Geomorphic processes operate within this nested hierarchy over a wide range of spatial and temporal scales; large-scale and long-term processes define stream system characteristics at the catchment level (external controls), whilst smaller-scale, short-term processes (internal controls) characterise morphology from the river segment scale down to the microhabitat level (Charlton, 2008).

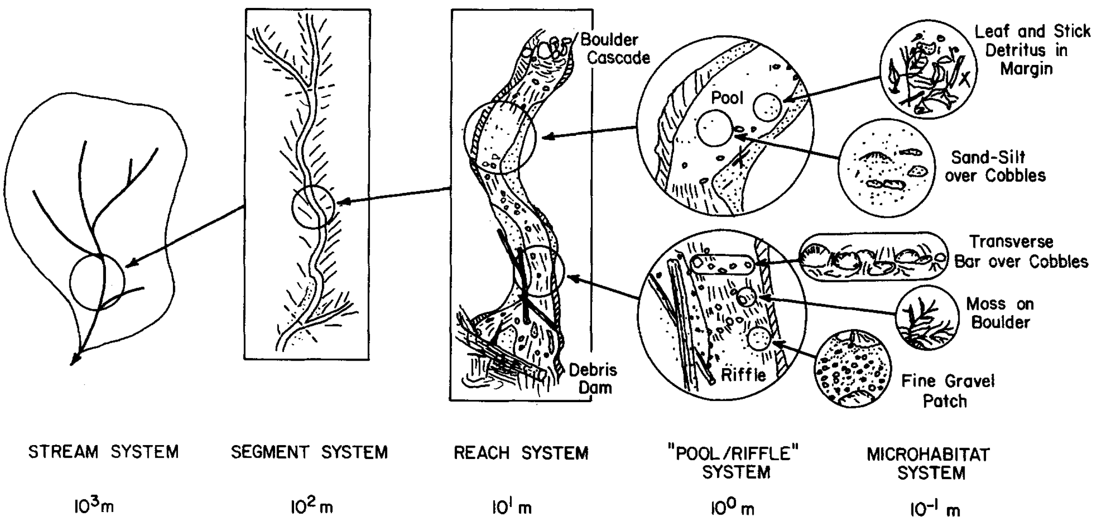



**Figure 1.3** Nested hierarchical organisation of a river and subsequent habitat systems. Hydrogeomorphic processes operate over variable spatial and temporal scales (from Frissell et al., 1986).

The relationship between water flow and sediment transport, and the associated geomorphic response, is fundamental to fluvial system processes (Leopold et al., 1964). Channel processes (erosion, transport and deposition) are controlled by the hydraulic regime and sediment supply, volume and grain-size distribution (Werritty, 1997; Easterbrook, 1999). The hydraulic regime is an important internal control on channel behaviour and modification. It has a seasonally variable character determined by climate (frequency and magnitude of precipitation), geological characteristics, catchment morphology, vegetation, channel characteristics and catchment land-use (Leopold et al., 1964; Charlton, 2008). Geological characteristics play a key role in the nature of channel discharge and the hydrological response to precipitation events; impermeable geology has little retention time resulting in greater channel peak flows, while permeable geology moderates channel flow maintaining base flows during the dry season (Sear et al., 1999). Stream power, the potential energy to impart physical channel form change, is determined by the channel gradient and discharge (Easterbrook, 1999; Charlton, 2008). An increase in either has an associated increase in energy and therefore potential geomorphologic activity (Easterbrook, 1999).

Sediment supply is spatially and temporally variable and determined largely by the frequency and magnitude of anthropogenic activities, catchment precipitation and the associated hydrograph, the nature of catchment processes, topography, geology, vegetation characteristics, volume and the grain-size distribution of available sediment as well as catchment disturbances (Easterbrook, 1999). Gradient, climate and channel boundary friction determine the transport potential of water (Leopold et al., 1964). A system imbalance occurs if sediment supply increases above the stream power threshold capacity to transport it resulting in deposition throughout the stream reach (Hey, 1996). The inverse results in accelerated channel boundary erosion. Deposition occurs where there is a reduction in velocity, a decline in slope angle, an increase in cross-sectional area (an effective decrease in velocity) and/or obstructions to flow (Charlton, 2008).

Erosion of very fine sediment such as clay requires great velocities to overcome the cohesive forces acting between like-sized particles (Nelson et al., 1987). These velocities are similar to those required to mobilise gravel (Figure 1.4). Once initial forces have been overcome downstream transport occurs at a reduced range of velocities. Channel form (width, depth, height, bedform wavelength, slope, sinuosity and meander wavelength) and the response mechanism are determined by channel processes (ultimately the hydraulic regime and the sediment supply) (Hey, 1997) but constrained by catchment geology, climate, stream power, vegetation and nature of channel processes (Leopold et al., 1964; Hey, 1997; Wharton, 2000). River channel form is naturally dynamic and constantly modified through the response to erosion, transportation and deposition, which in turn is defined by the relationship between stream power (slope and discharge) and the river channels' resistance to erosion (geology and vegetation) (Leopold et al., 1964; Gregory, 1992).

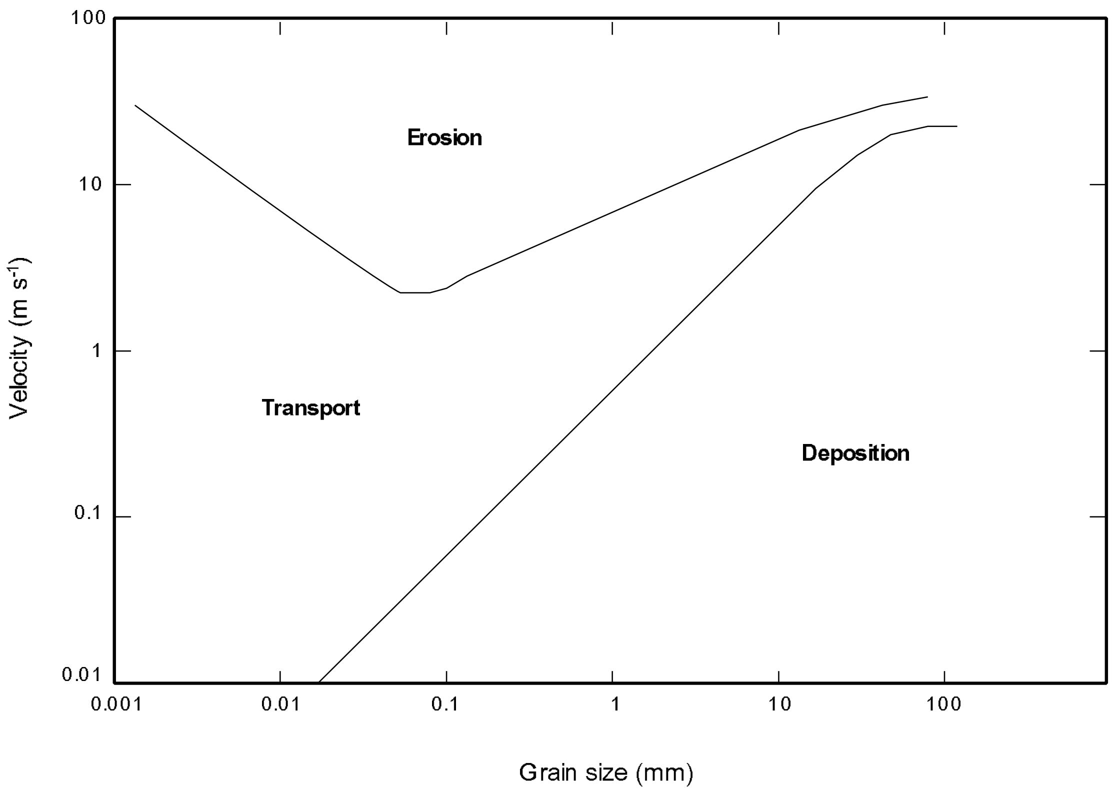



**Figure 1.4** Critical velocities for erosion, transport and deposition of sediments. Modified from Hjulström (1935). Relating sediment size into average velocity required for erosion, transportation and deposition, the Hjulström curve is simplistic in nature. Derived from flume based studies 1 m above the bed, the Hjulström curve fails to account for mixed sediment loads and does not include the complex parameters required to explain physical conditions under which such thresholds occur.

## 1.3 Anthropogenic impacts on fluvial systems

For little under 10 000 years humans settled within close vicinity of riverine environments to exploit the flat and fertile floodplains (Fagan, 1999). Ancient Egyptian farmers relied on rivers for irrigation of floodplain agriculture between 6000-3000 BC (Fagan, 1999; Wharton, 2000). Significant impacts to fluvial systems began with the large scale deforestation and removal of riparian vegetation for agricultural expansion from approximately 2000 BC (Eaton, 1989). Loss of catchment vegetation modified stream power through increasing sedimentation and catchment run-off rates, which increased channel discharge, altered hydrologic regimes and increased flood frequency (Eaton, 1989; Gregory, 1997). Furthermore, there was an associated reduction in marginal shading and an increase in temperature triggering excessive aquatic primary production (Eaton, 1989). Considerable reductions in allochthonous inputs and increases in autochthonous production (Webster et al., 1992) altered food-web function and structure.

Pressures on water resources intensified considerably with mid-19th century increases in global population associated with the Industrial Revolution and elevated rates of urbanisation (see Baer and Pringle, 2000). There was an associated intensification in channelisation and large drainage schemes, water meadow development and navigation canals (Wharton, 2000; Park, 2005a). The demands for food and manufactured products, flood protection, alternative power generation, inter-basin water transfers, effluent and waste product disposal exacerbated demands on water resource use (Boon, 1992). Economic growth of many countries have depended on such water resource exploitation (Mellquist, 1992). In order to sustain this level of exploitation rivers have been heavily regulated with hard engineered solutions to satisfy our needs. Anthropogenic modification has altered natural hydrogeomorphological processes, modifying channel behaviour and morphological character of fluvial systems. Land-use has greatly accelerated soil erosion and the supply of sediment to rivers with deleterious impacts on ecological components.

### 1.3.1 River engineering

River channelisation is the term used for the physical modification of river channels to make provision for human requirements, particularly flood mitigation (Brookes et al., 1983). Traditional river engineering assumed an environmental constancy over a geological and ecological irrelevant time scale (Macklin and Lewin, 1997; Sear, 1999). These approaches to flood management include one or a combination of the following:

1. *Re-sectioning* is the process whereby channel width and/or depth is increased to provide a greater channel area to accommodate increased discharges, frequently for flood water conveyance (Hey, 1996; Downs and Gregory, 2004). Re-sectioning frequently includes removal of channel features such as riffles, pools and depositional bars for greater flow efficiency, and limits channel connection with the floodplain (Wharton, 2000; Charlton, 2008).
2. *Re-alignment* is the process of channel straightening through removal of meander bends reducing the distance water travels, increasing gradient and thereby velocity with an associated increase in discharge rate (Hey, 1996). Realignment has historically canalised sections of river channel often associated with construction of a succession of navigation locks (Downs and Gregory, 2004).

3. *Dredging*, frequently done in combination with re-sectioning and realignment, is the process of increasing channel depth through the removal of bed substrate. Dredging reduces channel flow resistance, it increases velocity and removes bedforms such as pool-riffle habitat.
4. *Embankment and levee* construction increases the height of river banks restricting flow to the river channel and further reduces channel-floodplain connectivity (Hey, 1996; Wharton, 2000; Downs and Gregory, 2004). This is frequently done in combination with 3 above.
5. *Dams* and large weir structures regulate discharge (Downs and Gregory, 2004), altering channel processes and forming depositional environments that starve downstream reaches of sediment (Petts, 1984). Water flow is reduced upstream of a dam structure, increasing sediment deposition and water temperatures which effects the survival of salmonid embryos. Alternating water releases downstream of dams as a product of hydropower generation, can expose redds as water levels drop.
6. *Routine maintenance* maintains an open channel through removal of debris and natural structures that threaten to inhibit water conveyance. This involves the routine removal of large woody debris, large rocks, vegetation and sediment (Charlton, 2008).

### 1.3.2 Ecological consequences of river engineering

River channel engineering affects a change in system equilibrium altering the hydraulic regime and sediment transport processes throughout the catchment (Hey, 1996; Wharton, 2000; Charlton, 2008). Modification simplifies the natural physical complexity of the river channel. The biotic and abiotic components of the fluvial system are intimately connected (Giller and Malmqvist, 1998). Physical modification is therefore associated with a reduced community biodiversity and population abundance and has a marked impact on ecological functioning of fluvial systems (McCarthy, 1985). Traditional river engineering coupled with ecologically unsympathetic land-use altered fluvial system control variables and hydrogeomorphic integrity with subsequent adverse ecological implications.

Habitat diversity is fundamental to the continued persistence of biotic communities. Reduction in structural complexity and spatial fragmentation between life-stage dependent habitat has discernible implications for biodiversity (Mann, 1988). Channel modification has been cited as a leading cause behind declining fish abundance in lowland rivers (Spillet et al, 1985; Cowx et al, 1986; Swales, 1988). Bedforms, such as pool-riffle sequences, play a fundamental role in determining hydraulic functions vital to salmonid embryo survival, such as up/down welling, at a mesohabitat scale. Cowx et al. (1986) concluded that the loss of pool-riffle features had a marked reduction in fish abundance in the River Stour, Leicestershire. A study on the River Perry, a small channelised tributary of the River Severn, attributed the loss of habitat diversity with reductions of fish community diversity and population abundance (Swales, 1988). Loss of key habitat such as spawning gravels cause fish populations to be dominated by a few age classes creating an unstable population dynamic that frequently leads to severe declines in abundance (Mann, 1988). Loss of habitat can also induce a change of species composition (Mann, 1988; Swales, 1988). Such changes in composition have the potential to modify established ecological structure and function. Routine maintenance is frequently required in order to maintain the desired engineered state. Channel maintenance however leads to further instability and disrupts natural recovery processes (Swales, 1988; Brookes, 1992). Reduction of physical habitat and hydraulic refuges (Wolman and Schick, 1967; Swales, 1988, Pretty et al., 2003; Harrison et al., 2004), reductions in water quality and the associated decline in biodiversity (see Wads, 1995) generate an ecosystem with poor resilience.

### 1.3.3 A catchment based approach

Water resource management and river regulation is of significant socio-political, environmental and economical concern, particularly as pressures on water resources continue to increase whilst the quality decreases (Boon, 1992). A revision of the traditional approach to management has led to a more environmentally sensitive framework developed through greater understanding of the principles of hydrogeomorphology. Greater environmental awareness has over the past several decades led to an emphasis on a holistic and integrated management framework and the development of river rehabilitation (Gore, 1985).

The catchment, defined as an area of land drained by a single river network confined by hydraulic and topographic variables (Leopold et al., 1964), provides the basic geomorphic landscape unit for river management (Hooper and Margerum, 2000; Downs and Gregory, 2004). River management at this scale incorporates many different, often conflicting, resource demands and integrates fluvial system components at relevant hierarchal and functional scales (Downs and Gregory, 2004). Such a holistic approach has great sustainability and should ideally be incorporated into all development within the catchment (Gardiner and Cole, 1992). The challenge of management at this level is communication across a diverse range of scientific disciplines, statutory organisations, policy makers and stakeholders (Wade et al., 1995). River management at the catchment scale needs to develop restoration strategies based on sound scientific principles drawn from a diverse range of disciplines (Mellquist, 1992; Petts and Calow, 1996; Harper et al., 1999). In order to be effective in this role, Petts and Calow (1996) assert that land-use must be managed in a more sustainable manner. Catchment integrated river management requires co-operation between governmental and non-governmental bodies, statutory agreements, politicians, river managers, planners, scientists and local stakeholders in order to drive river rehabilitation into the 21st century. The Department for Environment, Food and Rural Affairs (Defra) initiated a Catchment Based Approach (CaBA) framework in 2013 to deliver river basin management planning in each of the stipulated Water Framework Directive catchments across England. CaBA recognises the integration of land and water at the natural catchment scale, and the importance of engagement and collaboration of stakeholders to integrate decision making and delivery of water improvement schemes. CaBA promotes development of suitable River Basin Management Plans, associated with the delivery of WFD objectives, through an interdisciplinary approach inclusive of local collaboration and decision-making. Development of formal Catchment Partnerships of key stakeholders, such as Rivers Trusts and Wildlife Trusts, recognised by the Environment Agency are key to the delivery of WFD objectives under the CaBA framework policy.

### 1.4 *Salmo trutta*: characteristics and life history review

*Salmo trutta* populations within coastal streams often exhibit polymorphism; the occurrence of both migratory and non-migratory morphs within the same river system (Näslund, 1995; Jonsson, 1989; Jonsson et al., 2001). The non-migratory population are sedentary, feeding and spawning in the river, whilst migratory fish perform these functions between the sea and freshwater. Environmental constraints such as low water levels and/or increased competition for limited resources such as habitat and food can in some cases cause non-migratory *S. trutta* to smolt (a physiological process in preparation for salinity osmoregulation) and migrate seawards (Elliot, 1994; Jonsson and Jonsson, 2009). They will usually feed in either estuarine or shallow coastal waters for between 1-4 years, before migrating back upstream to spawn (Jonsson and Jonsson, 1999; Jonsson et al., 2001; Klemetsen et al., 2003). Both non-migratory and migratory *S. trutta* depend on the same freshwater habitat for spawning and juvenile rearing.

### 1.4.1 Spawning and embryo development

Spawning typically commences in autumn and can continue though to early winter (Jones and Ball, 1954; Egglishaw and Shackley, 1977). Water temperature regulates the timing of *S. trutta* spawning (Moore et al, 2012). The timing within the same river is approximately similar, however, spawning varies regionally along a climatically-driven north-south divide in Britain (Menzies, 1936). Egglishaw and Shackley (1977) observed spawning between October and December in a Scottish stream, the Shelligan Burn, whilst *Salmo trutta* in chalk streams spawn between December and January (Mann et al, 1989).

*S. trutta* lay their eggs in river bed substrates where they are left to incubate with no parental care. The incubating gravel environment provides some protection for the vulnerable embryos from predation and the force of high water levels. Once the female finds suitable spawning gravels, the fish turns on her flank and, through a series of convulsion-like pulses, uses her tail to subject the gravel area beneath to vigorous agitation, displacing sediment into the water column (Jones and Ball, 1954; Hartman and Hakala, 2006; Marchildon et al., 2010). Stream velocity, turbulent mixing and sediment weight control the downstream sorting of the displaced substrate (Young et al., 1989; Greig et al., 2005a; Hartman and Hakala, 2006; Marchildon et al., 2010). The larger, heavier, substrate is not dislodged far relative to the finer material that is displaced further downstream. Eggs are deposited and fertilized in the resultant hollow (Jones and Ball, 1954; Greig et al., 2005a). The female fish then moves immediately upstream of this pit and excavates another hollow in a similar fashion, thus covering the previously deposited eggs with the displaced substrate (Figure 1.5). This process is repeated several times to produce a gravel mound or tailspill (commonly known as a redd) that contains the fertilised eggs (Jones and Ball, 1954).

The grain-size distribution of the tailspill differs from the surrounding gravel environment due to entrainment and wash-out of the finer sediment fraction during the redd cutting process (Kondolf et al., 1993; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006). Zimmermann and Lapointe (2005) observed that as much as 41% of fine sediment ($D$<2 mm) is displaced during the redd cutting process. Kondolf et al. (1993) reported a loss of 59.3% fines <1 mm from *S. trutta* redds. Similar results were obtained by Hartman and Hakala (2006) who reported significant differences of sediment <2 mm between redd substrate and non-redd substrate. The absence of fine material creates greater interstitial flow of water through the tailspill, delivering dissolved oxygen through infiltration to incubating embryos and removing toxic metabolic waste products through exfiltration.

Redd morphology plays a key role in driving localised hydraulic gradients into and out of redd substrate nested within larger scale pool-riffle river bed topography (Tonina and Buffington, 2009). Increased permeability and gravel depth around the egg pocket causes oxygen-rich water to downwell, whilst water upwells as gravel depth decreases in the tailspill (Vaux, 1968; Tonina and Buffington, 2009; Schindler Wildhaber et al., 2014). Further, the convex nature of the pit enhances downwelling into the egg pocket whilst concave morphology of the tailspill encourages upwelling out of redd substrate (Vaux, 1968). However, redd morphology contributes to localised hydraulic exchange processes over the short term only as redd morphology itself is altered as a result of fluvial conditions (Schindler Wildhaber et al, 2014).

Embryo development is temperature dependent. Ojanguren and Braña (2003) established that higher temperatures encouraged faster metabolic rates and tended to result in smaller alevins, whilst lower temperatures resulted in larger alevins. Lower temperatures during the early development stages also delayed emergence. In this manner the ambient temperature becomes a significant factor for the survival of emerged juveniles (Einum and Fleming, 2000). Ojanguren and Braña (2003) reported that the optimum temperature yielding maximum embryonic survival (from egg fertilization to exogenous feeding) was 8-10° C and mortality increased above and below this threshold, with an upper threshold limit of 14-16° C.

Moreover, embryo are susceptible to predation by leaches and fish. *Cottus gobio* (bullhead) predate on *S. trutta* embryo at a rate dependent on intragravel void sizes, the larger the void the grater the predation rate (Palm et al., 2009). Macroinvertebrates *Leuctra hippopus*, Ptychopteridae sp. and *Gammarus pulex* were observed to scavenge on trout embryo (Brown and Diamond, 1984). This observation lead Brown and Diamond (1984) as well as Ellis (1970) to conclude that macroinvertebrates were not predators of trout embryo but likely played an important role removing dead eggs from redd substrate. Further, salmonid embryos are susceptible to wash-out during high flow events (Lapointe et al, 2000; Fausch et al., 2001). Small increases in streambed scour (cm) to egg burial depths or mechanical damage and wash-out of embryos during bedload transport can have significant implications on embryo survival (Montgomery et al., 1996).

The eggs hatch into larvae (alevins) in spring. These feed for several weeks thereafter on yolk sacs attached to their undersides (Beer and Anderson, 1997). Both alevin (with remnants of yolk sac still attached) and fry (yolk sac completely resorbed) emerge from the incubating environment. The majority of fry emerge from redds within a short time period (8-18 days) mainly at night, with low levels of emergence for several hours to two weeks thereafter

(Elliott, 1986a; Moore and Scott, 1988). Alevin remain near bed substrata absorbing nutrients from their attached yolk-sac (Ottaway and Clarke, 1981). Once the yolk sac has been resorbed fry fill their swim bladders by engulfing air bubbles and attain neutral buoyancy (Elliott, 1986b) and begin feeding exogenously, a stage known as first-feeding (Klemetsen, 2003). Dispersal from bed substrata occurs at this stage (Ottaway and Clarke, 1981; Ottaway and Forrest, 1983) and feeding and refuge habitats are established. Juvenile fish are particularly vulnerable and this stage is characterised by high levels of mortality (Elliott, 1986b).

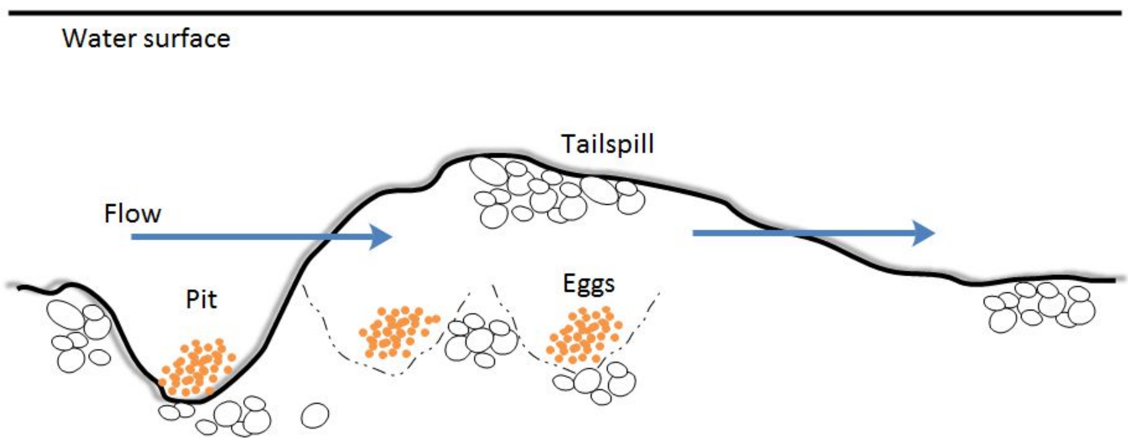



**Figure 1.5** Profile of a *S. trutta* redd during construction. Note several egg pockets and water flow through the tailspill.

### 1.4.2 The spawning habitat

Gravel-bed rivers, characterised by a surface armour layer of coarse sediment (256>*D*>2 mm) above mixed substrate, provide important spawning habitat for salmonid populations. Surface armour layers develop during sustained periods of low velocities where fine sediment material is entrained within the water column and winnowed out of a heterogeneous surface (Reid et al., 1997). This results in a greater proportion of surface coarse sediment relative to the underlying substrate. Surface armouring is usually a single particle diameter in thickness (Kondolf, 2000; Milan et al., 2000), typically preventing finer particles beneath from being entrained during high flows. As such a much greater shear stress threshold is required to remove the armour layer once formed (Reid et al., 1997; Charlton, 2008).

The composition and structure of streambed substrate play a vital role in sediment processes within gravel deposits (Reid et al., 1997). Bed substrate will erode at some equal flow condition, whilst poorly sorted substrate require a range of flows. The quality, distribution and size of spawning substrate are controlled by upstream hydrogeomorphic processes and catchment characteristics resulting in considerable spatial and temporal variability (Kondolf and Wolman, 1993; Milan et al., 2000; Hendry et al., 2003; Greig et al., 2005a; Moir and Pasternack, 2010). *S. trutta* populations are a function of the accessibility and suitability of spawning gravels within a given body of water (Kondolf and Wolman, 1993; Kondolf, 2000).

Braided, meandered and straight channel gravel-bed rivers develop quasi-stable riffle-pool bedforms (Leopold et al., 1964. Thompson, 1986). Riffle-pool sequences are morphological characteristics typically associated with gravel-bed streams. Pools are topographical depressions formed through bed scour and characterised by relatively fine bed material, whilst riffles are topographically high produced by deposition of coarser bed material (Keller, 1971). The regular pattern of scour and deposition required for the mechanics of riffle-pool formation come from alternate convergent and divergent flow patterns (Yalin, 1971; Richards, 1976) which, in turn, are produced by a meandering thalweg (Keller, 1972). A meandering thalweg is produced in a straight channel by convergent cells of secondary flow induced by wall turbulence (Einstein and Shen, 1964). Water flow is non-uniform and spatially variable creating alternate zones of gravel accumulation within the channel that develops into equally spaced deposits approximately 5-7 channel widths apart interspersed by areas devoid of gravel (Leopold et al., 1964; Folkard, 2005b). The resulting riffle-pool sequence further develops channel topography and asymmetry through lateral deposition resulting in reinforcement of secondary flows which initiate meander development through positive feedback (Thompson, 1986; Folkard, 2005b). In this manner riffle-pool morphology is an integral feature in the mechanics of meander development (Leopold et al, 1964; Thompson, 1986).

The maintenance of a riffle-pool morphology is somewhat contentious, although the hydraulic reversal hypothesis is the most accepted. Gilbert (1914) was the first to describe a reversal in near-bed velocity in riffle-pool morphology. However, Keller (1971) was credited with first quantifying velocity reversal. The velocity reversal hypothesis states that near-bed velocity is lower in pools than in adjacent riffles, but with an increase in discharge, typically associated with flood events, near-bed velocity in pools increases faster than adjacent riffles (Keller, 1971). This hydraulic reversal prevents sediment infilling of pools due to the high energy associated with peak flood hydrographs. At low flows riffles are characterised by greater near-bed velocity gradients and erosion of fine sediment that is deposited in pools, whilst at high flows velocity is reversed and coarse material is scoured from pools and deposited in riffles (Keller, 1971; Robert, 1997). In this manner riffle-pool morphology is maintained by a reversal of hydraulic processes (near-bed velocity and shear stress) induced by high flow events. Robert (1997) observed that the difference in near-bed velocities between pools and riffles decreased as discharge increased, thus suggesting a hydraulic reversal at or near bankfull discharge. Sawyer et al. (2010) observed riffle-pool maintenance through velocity reversal at the peak flood hydrograph, consistent with Keller (1971). This study, however, failed to evaluate long-term riffle-pool persistence as it focused on a single flood event. Milan et al. (2001) reported consistency with Keller's (1971) velocity reversal theory through river stage dependent variability in shear stress throughout a riffle-pool sequence. However, velocity reversal was considered marginal by the authors as it was not evident in all individual riffle-pool units. Heritage and Milan (2004) proposed that hydraulic reversal only occurred for short periods of time during the peak flood hydrograph and as such was not able to transport all available sediment. Hydraulic reversal requires material to be transported between pool and riffle features in quasi-equilibrium to ensure pool infilling does not occur. However, the authors observed an imbalance of energy exerted at the riffle unit and as such concluded that reversal failed to explain the excess energy available (Heritage and Milan, 2004).

Pressure gradients developed around geomorphological junctions, such as changes in water depth between pool and riffle bed forms, drives surface-subsurface water exchange (Harvey and Bencala, 1993; Brunke and Gonser, 1997). In their laboratory flume experiments of hyporheic exchange in gravel-bed pool-riffle channels, Tonina and Buffington (2007) used a three-dimensional pumping exchange model, FLUENT 6.0, to demonstrate that exchange was driven by bedform advection. Hyporheic exchange in their experiments was shown to vary based on the complex interaction between discharge and bedform topography. An increase in discharge and bedform topography lowered hyporheic penetration depths (decreasing hyporheic exchange). However, Tonina and Buffington (2007) noted that hyporheic exchange is not exclusively dependent on discharge and that groundwater slope and alluvium depth are likely important factors. River channels might have several zones of hydrological downwelling and upwelling dependent on substrate permeability, hydraulic conductivity and catchment topography (Brunke and Gonser, 1997; Malard et al., 2002). Downwelling characteristically occurs at the upstream end of riffles whilst upwelling at the downstream end (Brunke and Gonser, 1997; Malard et al., 2002). This is important for embryo development as well as the cycling and storage of nutrients and organic matter (Findlay et al., 1993; Findlay, 1995; Jones et al., 1995; Triska et al., 1993). Decomposition of organic compounds in the hyporheic zone mineralizes carbon to carbon dioxide and nitrogen to ammonium ($NH_4^+$), which then oxidises to nitrate (Valett et al., 1994; Jones et al., 1995). Surface-subsurface water exchange provides important nutrient cycling and could be either a sink or source of nutrients within streams (Valett et al., 1996). Furthermore, upwelling water is cooler than the ambient surface water in summer and warmer in winter maintaining annual stream temperature stability. The hyporheic zone connecting groundwater and river water is therefore important for stream ecosystem functioning (Storey et al., 1999).

Water depth, velocity, substrate size and composition are considered the most important determining variables for salmonid spawning habitat (Armstrong et al., 2003). *S. trutta* are known to spawn in water depths ranging between 15-60 cm, within a velocity range 0.20-0.75 m $s^{-1}$ and on substrate size of $8 \geq D \geq 64$ mm with little fine sediment $D<1$ mm (Jutila, 1992; Kondolf and Wolman, 1993; Armstrong et al., 2003; Louhi et al., 2008; Marchildon et al., 2010). Milan et al. (2000) proposed that egg burial depth may be controlled by the spawning substrate and extent of gravel depth. However, our knowledge of spawning parameters is limited by constraints of safe working conditions in the field (Marchildon et al., 2010). Many researchers agree that *S. trutta* display a degree of elasticity in spawning habitat selection determined by gravel availability and spawning competition (Witzel and MacCrimmon, 1983; Kondolf and Wolman, 1993; Barlaup et al., 2008; Marchildon et al., 2010). The upper limit of gravel size and velocities in which *S. trutta* are able to spawn is directly correlated with body size; larger fish can use greater gravel sizes and spawn in higher velocities and therefore have a competitive spawning advantage (Witzel and MacCrimmon, 1983; Crisp and Carling, 1989; Kondolf and Wolman, 1993; Moir and Pasternack, 2010). Substrate size also effects embryo development. Substrate with a low $D_{50}$ (median gravel diameter) has earlier emergence times than substrate with a greater $D_{50}$, resulting in a preponderance of smaller fry with partially resorbed yolk sacs (Rubin, 1992). Smaller fry are less able to compete for the more favourable habitat, hold station against the current and get displaced further than their larger cohorts.

Spawning substrate are characterised by both larger grained framework gravels and cobbles, which provide support to the deposit, and finer grained matrix sediments that fill the interstices between framework gravels (Kondolf and Wolman, 1993; Milan et al., 2000). The relative contribution of matrix to framework sediments characterise the suitability of a spawning habitat. High quality spawning gravels are framework gravel supported with a low percentage of matrix sediment (Lotspeich and Everest, 1981; Acornley and Sear, 1999; Kondolf, 2000; Milan et al., 2000; Greig et al., 2005a; Hartman and Hakala, 2006; Louhi et al., 2008; Harvey et al., 2009). Greater water velocities have an increased propensity to flush fine sediments from spawning substrate, creating a framework-supported bed (Milan et al., 2000).

These higher flows have more energy with which to transport both framework and matrix sediments, creating loose and fine sediment free spawning habitats. Low energy streams, however, cannot easily mobilise framework gravels and transport mainly matrix sediments (Milan et al., 2000; Merz et al., 2004). This predisposes spawning substrate in these streams to surface armouring (Milan et al., 2000).

The requirement to assess spawning gravel quality was identified early in the history of salmonid research and management (Harrison, 1923 cited in Lotspeich and Everest, 1981). Lotspeich and Everest (1981) proposed the use of a single-variable index to assess the quality of potential spawning gravel, the Fredle Index $f_i$ (equation 1), calculated by dividing the geometric mean particle diameter size ($d_g$) by the sorting coefficient ($S_o$), a measure of dispersion (equation 2):

Equation 1.1:

$f_i = d_g/S_o$

Equation 2.1:

$S_o = [(d_{75})/(d_{25})]^{0.5}$

where $d_{75}$ and $d_{25}$ represent the grain-sizes at which 75% and 25% of the size distribution are finer. Using these properties of the grain-size distribution, the Fredle Index provides an indication of gravel permeability and pore space. Lotspeich and Everest (1981) advanced this index in light of the absence of an agreed standard method to assess the quality of spawning gravels. Use of the Fredle Index was later criticised as an oversimplification, with wider recognition that gravel size requirements were not constant throughout the spawning process and varied depending on reproduction stage (Crisp, 1993; Kondolf and Wolman, 1993; Kondolf, 2000; Milan et al., 2000). During the redd cutting process, framework gravels should be easily moved by the female fish, the upper limits of which are dependent on fish length, water velocity and degree of gravel embededness (Crisp and Carling, 1989; Kondolf and Wolman, 1993). Successful embryo incubation within the redd requires low concentrations of fine sediment (Lotspeich and Everest, 1981; Kondolf et al., 1993; Hartman and Hakala, 2006; Louhi et al., 2008; Harvey et al., 2009).

Typically, sediment <1 mm is considered detrimental to salmonid spawning as desposition reduces interstitial permeability (McNeil and Ahnell, 1964; Cederholm and Salo, 1979; Tagart, 1984). Streambed gravels with reduced quantities of sediment <1 mm therefore approach optimum habitat for spawning (Kondolf, 2000). However, in a gravel cleansing experiment to improve hyporheic water quality, Meyer et al. (2008) found that reduction of sediment <2 mm to below 0.2% of the total sediment composition significantly improved DO concentrations. Simulated redds constructed with homogonous gravel sizes 1.5, 4.8, 9.6, 18.0 and 32.0 mm indicated that redds constructed with gravel sized 18.0 mm had the greatest embryo survival and alevin emergence (Olsson and Persson, 1986). A significantly lower survival rate was observed from redds constructed with smaller grain-sizes 1.5 mm and 4.8 mm (Olsson and Persson, 1986). Olsson and Persson (1986) concluded that it was likely that embryo mortality occurred early in the development stage in redds with smaller grain sizes (1.5 mm) as permeability was reduced, and hence oxygen delivery repressed, relative to those redds of larger grain-sizes. A small composition, 1.5%, of clay and silt sized particles (<0.125 mm) within spawning substrata significantly impact oxygen uptake by developing embryos by blocking micropore canals in the chorion (Lapointe et al., 2005, Greig et al., 2005b, Julien and Bergeron, 2006, Levasseur et al., 2006).

Kondolf (2000) argued that the geometric mean particle size can be similar for very different grain-size distributions and is dependent on the relative proportions of fine to large sediments. The use of the full grain-size distribution is therefore preferred to any single descriptor index (Lapointe et al., 2004).

### 1.4.3 Effects of fine sediment on embryo development

Fine sediment accumulation in redd substrate is a variable natural process controlled by hydrogeomorphological as well as biological factors (Hendry et al., 2003; Greig et al., 2005a). Anthropogenic inputs accelerate sedimentation rates. Diffuse run-off associated with arable land-use, road construction and tree felling within the catchment significantly increase inputs of fine sediment and organic waste products to the river system (Walling, 1995; Theurer et al., 1998; Hendry et al., 2003; Greig et al., 2005a; Zimmermann and Lapointe, 2005). The European-wide change in cultivation from spring to autumn sown crop varieties over the past 50 years and the intensification of agricultural practice encouraged by the subsidy scheme offered to farmers by the European Common Agricultural Policy (CAP) introduced in 1962 has likely further intensified the rate of siltation (Hendry et al., 2003; Greig et al., 2005a).

Although a significant amount of fine sediment is displaced during the redd cutting process by entrainment within the water column (Kondolf et al., 1993; Zimmermann and Lapointe, 2005;

Hartman and Hakala, 2006; Marchildon et al., 2010), excessive accumulations of fine sediments have a detrimental effect on the survival of *S. trutta* embryos during incubation. Interstitial gravel spaces enable oxygen-rich water to flow through the incubation sediments and thereby sustain the gradient required to drive diffuse oxygen exchange across the egg membrane during embryonic development (Greig et al., 2005a; Greig et al., 2005b). Intrusion of fine sediment during embryo development inhibits gravel permeability and consequently the delivery of oxygen to the developing egg (Theurer et al., 1998; Greig et al., 2005a; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006). Indeed, interstitial velocities can be significantly reduced by a single sediment run-off event (Zimmermann and Lapointe, 2005). Furthermore, a reduced interstitial flow is less able to remove the toxic metabolic waste products associated with embryonic development (Burkhalter and Kaya, 1975). Agricultural run-off typically contains organic and nutrient-rich sediment (Greig et al., 2005a). Delivery of these compounds into redd gravels encourages algal growth that further exacerbates oxygen demand and uptake (Olsson and Persson, 1986; Greig et al., 2005a). Greig et al. (2005b) noted that clay particles ($D$<4 μm) physically blocked membrane micropore canals, significantly reducing the efficient exchange of oxygen across egg membranes thereby inhibiting embryonic growth. Oxygen demand varies with embryonic developmental stage and ambient water temperatures (Louhi et al., 2008). Therefore accumulation of fine sediment will have variable effects on incubating embryos at different stages of growth for different temperature conditions. Sediment accumulation affects embryo growth both directly and indirectly through a complex interaction between interstitial permeability, oxygen availability, temperature and groundwater upwelling (Greig et al., 2005a).

Whether suspended sediment is excluded, trapped or accumulated within embryo incubation substrate is determined by the ratio between interstitial pore size and suspended sediment size (Frostick et al., 1984; Lisle, 1989). Moreover, this ratio determines whether particles settle in surface sediments or accrue within deeper substrate (Lisle, 1989). The greater the size difference the more susceptible incubation gravels will be to fine sediment accrual within deeper substrate. Streambed framework particle shape has a further effect on sediment accrual. Increased roundness has greater accrual as porosity is more consistent than it might be with more angular shaped particles (Lisle and Eads, 1991). Well sorted spawning gravels in an environment with high suspended sediment loads are particularly vulnerable to deposition. These sediments will typically accumulate bottom-up, reducing permeability and consequently intragravel velocity (Greig et al., 2007).

The size of deposited material has variable effects on redds and the incubating embryos. Finer sediments filter through the upper redd substrate, decreasing interstitial spaces from the bottom-up (Einstein, 1968; Turnpenny and Williams, 1980; Acornley and Sear, 1999). Additionally, these finer sediments are more readily flushed from the upper levels during periods of increased discharge (Acornley and Sear, 1999). During the final stages of embryo development alevins emerge from redd gravels via intragravel pore spaces. However, Beschta and Jackson (1979) noted that sand ($D_{50}$ = 0.5 mm) tended to settle in the upper 10 cm of a stable gravel bed under laboratory conditions. In this manner sands form a physical barrier that inhibits the passage of alevins during emergence (Crisp, 1993; Kondolf, 2000; Hartman and Hakala, 2006). Consequently, finer sediments have a detrimental impact on embryo development, whilst larger sand-sized sediments impair the later swim-up stage.

Olsson and Persson (1986) observed a positive correlation between alevin length and weight against gravel size. Greater concentrations of finer gravel (1.5 and 4.8 mm) within redds tended to encourage premature emergence of predominantly smaller alevins with larger yolk sacs than those redds with a greater concentration of larger gavel (18 and 32 mm). The greater the peat concentration (60%) within redds the lower the survival and emergence (Olsson and Persson, 1986). Alevins with larger yolk sacs are poor swimmers and are therefore at greater risk of predation. Finer sediment (<2 mm) and greater organic material concentrations (40% and 60% respectively) are associated with higher embryo mortality early in the egg development stage due to a lack of dissolved oxygen, whilst coarser sediment sizes had higher mortalities at the alevin yolk sac stage (Olsson and Persson, 1986). Jensen et al. (2009) reanalysed previously published data and constructed probability models to examine the relationship between egg mortality rates and increasing fine sediment concentrations during incubation. A complex nonlinear relationship predicting a 1% increase in fine sediment ($D$<0.85 mm) resulted in a 17% reduction in survival (Jensen et al., 2009). Accumulations of fine sediment ($D$<1 mm) reduce interstitial permeability with subsequent detrimental effects on egg survival (Kondolf, 2000; Hartman and Hakala, 2006). For example, Hartman and Hakala (2006) found that significant decreases of juvenile *S. trutta* were correlated with minor increases in very fine sediment ($D$<0.063 mm) and suggested that threshold limits of sediment ($D$<0.063 mm) in spawning gravels should not exceed 1%. Quantification of fine sediment abundance (using a predetermined size threshold) is consequently an indirect but cost effective and reliable manner in which spawning gravel habitat quality can be assessed.

Suspended sediment concentration levels under current land-use have a key role in the deterioration of spawning gravel beds (Carling, 1984). In a study of the River Test, Hampshire, Acornley and Sear (1999) established seasonal suspended sediment concentrations and associated deposition rates. During summer, suspended sediment concentrations were in the region of 5 mg $l^{-1}$, but were much higher during winter, averaging over 20 mg $l^{-1}$. Annual concentrations ranged from 1-225 mg $l^{-1}$. Some 96% of the annual suspended sediment load was mobilised during the salmonid incubation period (Acornley and Sear, 1999). The composition of suspended sediment was mostly silt ($D$ = 4-63 µm), although sand ($D$ = 63-250 µm) entered suspension during higher discharge stages. Input of organic material varied seasonally; 25-40% during summer flows, and 15-25% during winter discharges. Deposition varied temporally and spatially, with greater deposition during winter peak discharge events: 0.02 kg $m^{-2}$ $day^{-1}$ during summer to 0.5-1.0 kg $m^{-2}$ $day^{-1}$ during winter (Acornley and Sear, 1999). Using these rates of deposition Acornley and Sear (1999) estimated that it would take approximately 25 days for the grain-size distribution of a freshly cut redd to return to original levels prior to displacement in the River Test. In comparison, suspended sediment concentrations observed in upland streams range between 0.80 mg $l^{-1}$ to 818 mg $l^{-1}$, consisting largely of coarse silt with a mean grain-size 0.017-0.041 mm (Carling, 1983). Coarsening of the suspended sediment load was associated with an increase in stream competency observed during peak flood discharge (Carling, 1983).

### 1.4.4 Recently emerged and first-feeding juvenile stages of *S. trutta*

The juvenile life-stage refers to the alevin, fry and parr (fry at several months) stages during the first year of life. Natural mortality rates of the recently emerged and first-feeding alevin and fry stages are greater than at any other life-stage (Egglishaw and Shackley, 1977; Mortensen, 1977b; Elliott, 1986b; Vetter, 1987) and dependent on habitat availability and quality (Egglishaw and Shackley, 1977). Newly emerged fry select suitable habitat based on velocity and water depth (<40 cm) parameters (Bohlin, 1977; Heggenes et al., 1999; Heggenes, 2002). Velocity is a key determinant of growth rates and population density (Bachman, 1984) with snout velocities (at level of fish head) <0.20 m $s^{-1}$ (Heggenes et al., 1999). Post emergent fry seek hydraulic refuges either along stream margins or close to spawning substrate (Solomon and Templeton, 1976; Elliott, 1986b) seeking refuge within interstices (Heggenes and Traaen, 1988; Bardonnet and Heland, 1994; Bardonnet et al., 2006).

Elliott (1989) proposed a critical survival period of between 25-70 days after emergence in which fry need to establish feeding and refuge habitats. Beyond this window those fry which do not establish suitable refuges do not contribute to population recruitment (Elliott, 1986b). Displaced fry are more vulnerable and at risk of increased mortality (Elliott, 1986b; Crisp and Hurley, 1991). Heggenes and Traaen (1988) established a critical velocity of 0.1 to 0.25 m $s^{-1}$ at which downstream displacement occurred. Post emergent fry displacement is primarily associated with stream velocity (Ottaway and Forrest, 1983; Elliott, 1987; Heggenes and Traaen, 1988; Moore and Scott, 1988; Crisp and Hurley, 1991) however, juvenile developmental stage (Ottaway and Clarke, 1981; Daufresne et al., 2005), competition for habitat space (Daufresne et al., 2005), predation and density of emergence are important determinants (Elliott, 1986b).

The size of fry is a survival advantage. Larger fry can go longer without feeding, have a greater intra-cohort competitive advantage, a reduced predation risk and are less vulnerable to hydraulic conditions (Ottaway and Forrest, 1983; Heggenes and Traaen, 1988; Elliott, 1989). Smaller alevins, with yolk sac still attached, establish hydraulic refugia within substrate interstices and are therefore not displaced far, while larger exogenously feeding fry need to enter the water column to feed and are susceptible to displacement under stream velocity (Ottaway and Clarke, 1981). Fry mortality during exogenous feeding is density-dependent and associated with elevated competition for habitat (Mortensen, 1977a; Mortensen, 1977b). Territorial and solitary behaviours are evident during these early life-stages (Elliott, 1986a). Smaller individuals are outcompeted from more favourable habitat by larger more dominant conspecifics forming a size-quality hierarchy of habitat use (Chapman, 1966; Bachman, 1984; Greenberg, 1994; Heggenes, 2002). Low habitat abundance can therefore increase intra-cohort competition generating density-dependent habitat use that affects carrying-capacity (Bohlin, 1977; Armstrong and Griffiths, 2001). As fry develop into parr they illustrate a preference for deeper habitat, approximately 10-60 cm with an associated increase in velocities, 0.05-0.5 m $s^{-1}$ (Egglishaw and Shackley, 1982). Suitable juvenile habitat is therefore associated with population density (Mortensen, 1977b). Habitat availability and the associated intra-cohort interaction regulate salmonid population production at the juvenile life-stages (Egglishaw and Shackley, 1977; and see Elliott et al., 1997).

### 1.4.5 Over-wintering habitat parameters of *S. trutta*

Peak fry migration occurs in autumn as *S. trutta* move greater distances to establish suitable over-wintering refuge habitat (Elliott, 1986a; Cunjak and Power, 1987; Bunnell et al., 1998; Homel and Budy, 2008). Shallow (<10 cm from bed substrate) (Heggenes, 2002), low velocity (0.1-0.3 m $s^{-1}$) areas (Cunjak and Power, 1986; Knouft and Spotila, 2002) in either substrate interstices (Heggenes and Saltveit, 1990; Greenberg, 1994; Heggenes and Dokk, 2001) or amongst vegetation and woody debris (Harvey et al., 1999; Heggenes et al., 1999) are suitable. Fry are sedentary during the winter months (Valdimarsson and Metcalfe, 1998; Brown et al., 2001; Dare et al., 2002) with small home ranges, moving to and from feeding habitat at night (Heggenes et al., 1993; Brown et al., 2001; Heggenes and Dokk, 2001; Griffiths et al., 2002). Sharing of over-wintering habitat is uncommon (Armstrong and Griffiths, 2001; Harwood et al., 2001; Griffiths et al., 2002). Intra-specific competition increases as habitat availability diminishes (Harwood et al., 2001; Griffiths et al. 2002) and as such habitat availability is population density-dependent (Meyer and Griffith, 1997). Egglishaw and Shackley (1977) and Elliott (1986) associated declines in abundance to in-stream mortality as opposed to loss of abundance due to migration. The availability of over-wintering habitat therefore regulates salmonid production during winter (Egglishaw and Shackley, 1977; Meyer and Griffith, 1997; Cunjak et al., 1998; Armstrong and Griffiths, 2001; Lund et al., 2003). However, physical and biological parameters (Ultsch, 1989; Cunjak and Therrien, 1998) as well as localised climatic conditions (Needham et al., 1945) are important population stresses that also contribute to mortality rates during this time.

### 1.4.6 Spatial relationship between critical life-stage dependent habitat

*Salmo trutta* are relatively sedentary and normally do not move extended distances throughout the river channel during the growth season (spring to autumn) (Egglishaw and Shackley, 1977; Bachman, 1984; Armstrong et al., 1994). Both Bunnell et al. (1998) and Knouft and Spotila (2002) observed that distance moved over a 24 hour period was associated with increasing body size. Juvenile *S. trutta* undergo two distinct life-stage dependent habitat relocations. The first is the distribution of recently emerged fry from spawning substrate to nursery habitat (velocity: 0-0.2 m.s-1, depth: 50-300 mm, grain-size: 10-90 mm) where they remain until the approach of winter (Jonsson, 1989; Greenberg, 1994; Heggenes et al., 1999; Armstrong et al., 2003; Hendry et al., 2003). The distribution distance of fry from spawning gravels to nursery habitat is density-dependent and associated with available habitat resources (Elliott, 1986b). A second migration to suitable over-wintering refuge habitat (grain-size: 60-500 mm, low degree of surface armouring in substrate, dense brushy margins, access to deeper and slower water) is observed as winter approaches (Solomon and Templeton, 1976; Alfredsen and Tesaker, 2002; Annear et al., 2002; Armstrong et al., 2003). Key life-stage habitat relocations have been observed to be mostly (>90%) in a downstream direction in chalk streams (Solomon and Templeton, 1976; Moore and Scott, 1988) while migration in non-chalk streams is not predisposed to any specific direction (Armstrong et al., 1994). The relative abundance of suitable habitat found in chalk streams has been identified as the dominant factor governing juvenile migration direction (Solomon and Templeton, 1976; Armstrong et al., 1994).

Not only is the availability and suitability of critical life-stage dependent habitat essential for salmonid production, but the spatial relationship between habitat and the ability to disperse between them a fundamental determinant of salmonid production at the juvenile stage (Kocik and Ferreri, 1997; White, 1999). Salmonid production can therefore be constrained through inadequate access to suitable physical habitat prompting high levels of inter- and intraspecific competition and predation (de Jalón, 1995). Greater production occurs in stream reaches of greater habitat density where suitable foraging and refuge habitat are in close proximity (Kocik and Ferreri, 1997). Spatial relationships between the habitat required for different life-stages of salmonids and the ability to disperse between them is critical to our understanding of the natural spatial scale at which to investigate population production (Kocik and Ferreri, 1997).

### 1.5 Flow biotopes and functional habitat

Interdisciplinary research between geomorphology, hydrology and river ecology (Harvey and Clifford, 2009) driven by the requirement to assess river habitat quality through field-based surveys developed in response to satisfying international legislation such as the EU Habitats Directive, and more recently the WFD, lead to the development of the physical or 'flow' biotope concept. A biotope is a region or area defined by homogeneous environmental conditions (Park and Allaby, 2013; Allaby, 2014). Flow biotopes are therefore units of largely homogonous flow characterised by hydraulically distinct surface flow types (Padmore, 1998) (Table 1.1). Studies conducted by Jowett (1993) and Wadeson (1994) indicated that surface flow type and degree of turbulence were described by Froude Number and as such was used to discriminate between flow biotope types. Surface flow, however, can be spatially and temporally variable with alternating flow stage (Clifford et al., 2006; Harvey and Clifford, 2008), and secondary flow types within primary or dominant flow types often under-represented (Padmore, 1997).

Flow biotopes are controlled by in-stream physical processes and as such form distinct habitats, observed by vegetative and minerogenic characteristics (Jowett, 1993; Wadeson, 1994; Padmore, 1997; Newson et al., 1998). Key descriptive river channel features such as riffles, pools, runs and glides are hydraulically defined units of flow (Wadeson, 1994). These functional habitats are defined units of distinct biotic assemblages (Armitage et al., 1995; Pardo and Armitage, 1997). Studies linking flow biotopes with functional habitats provide ecological context to flow biotope assessments (Harper et al., 2000; Harvey et al., 2008). Kemp et al. (2000) observed an association between flow biotope and functional habitat using Froude Number. Characterisation of flow biotopes are therefore ecologically valuable and make inferences about, and are a measure of, biotic diversity (Townsend and Hildrew, 1994; Harper et al., 2000). Visual identification of flow biotopes are considered statistically valid and a suitable standard unit of instream assessment (Newson and Newson, 2000). Flow biotopes have therefore been used to assess river habitat at the mesoscale, and consequently biotic diversity, and have been integrated into the UK River Habitat Survey (RHS) (Padmore, 1997 and 1998). The RHS was developed by the National Rivers Authority (NRA), and still used by the EA to record physical features of rivers at a national level (Raven et al., 1997; Raven et al., 2000; Harvey et al., 2008). This method enables rapid and reproducible surveys based on the concept of flow biotopes. Identification and assessment of flow biotopes has practical application for river rehabilitation based on the understanding that biological communities are fundamentally linked to and defined by physical variables (Statzner et al., 1988; Heede and Rinne, 1990).

**Table 1.1** Flow biotope and associated flow types, modified from Newson et al. (1998) and Newson and Newson (2000).

| Flow Biotope | Flow type |
|---|---|
| Waterfall | Free fall: vertical fall without obstruction from distinct feature |
| Spill | Chute: fast, smooth boundary turbulent flow over boulders/bedrock |
| Rapid | Broken standing waves: white water present |
| Cascade | Broken, standing waves: white water 'tumbling' waves, crest upstream facing |
| Riffle | Unbroken standing waves: undular standing upstream facing waves |
| Run | Rippled: no waves, disturbed rippled surface |
| Boil | Upwelling: heaving water, visible upwelling breaks surface |
| Glide | Smooth boundary, turbulent flow: perceptible smooth downstream movement, low roughness |
| Pool | Scarcely perceptible flow, full channel width: no net downstream flow |
| Deadwater | Scarcely perceptible flow, not full channel width: associated channel margins |

## 1.6 River rehabilitation

The lack of scientific understanding underpinning traditional approaches to river management has caused loss of ecological integrity and geomorphological instability (Mellquist, 1992). The concept of river rehabilitation has progressed out of greater environmental awareness acknowledged as the basis for sustainable development and the increased responsibility of legislative bodies for environmental protection (Sear, 1994). Restoration infers a full structural and functional return to a pristine previous state (Bradshaw, 1996; Holmes, 1998; Downs and Gregory, 2004). Defining complete pre-disturbed reference conditions is challenging, however, since the natural conditions of many of our rivers is mostly unknown (Wade et al., 1998). As such restoration in this sense is hard to achieve and is not a particularly realistic objective (Bradshaw, 1996). Rehabilitation refers to the partial structural and/or functional reinstatement to some former non-pristine state (Wade et al., 1998; Downs and Gregory, 2004). Rehabilitation therefore offers more feasible and achievable objectives. To date Boon (1998) contends that in fact most restoration schemes can be more accurately described as rehabilitation. Subsequently the term rehabilitation is preferred and will be used throughout the rest of this thesis. The challenge of rehabilitation lies in integrating knowledge of the interactions between biota, their physical environment and the prevailing hydrogeomorphological components whilst maintaining essential user requirements (Mainstone and Holmes, 2010). Thus integration of a wide range of disciplines throughout the rehabilitation processes is required (Wade et al., 1998). The extent of rehabilitation schemes should be determined by the nested hierarchal scale of spatial and temporal habitats (Harper et al., 1999).

Salmonids are sensitive to environmental change, have great economic value and are an iconic fish. As a result they have been the subject of much academic investigation. One of the essential objectives for the UK Biodiversity Action Plan (BAP) is to restore the natural range of native fish species (UK Steering Group, 1995a). Flow regulating structures such as weirs, tidal gates and mill structures hinder, and in many instances prevent, migrating salmonids from reaching their upstream spawning grounds and other key life-stage dependent habitat. Additionally, flood defence schemes, poor land drainage management and excessive accumulations of fine sediment have been largely responsible for the decline of global salmonid populations through the loss of suitable spawning and juvenile rearing habitat (Acornley and Sear, 1999; Hendry et al., 2003; Pedersen et al., 2009). Recognition of these barriers to population growth has led, within recent decades, to restorative measures and more recently governmental policy advocating the rehabilitation process (Hendry et al., 2003; Merz et al., 2006).

### 1.6.1 Legislative background to river rehabilitation

Several European Union (EU) Directives set out the statutory framework for acceptable levels of river resource use that are mandatory for national legislation by member states (Iversen et al., 2000). Of these the Water Framework Directive (WFD) (2000/60/EC) is the most significant and its objectives are currently the greatest driver for river rehabilitation in the European Union. The WFD provides an Integrated River Basin Management framework for Europe requiring member states to assess surface waters for good ecological status through ecological, morphological and chemical parameters (CEC, 2000). In addition the WFD requires member states to develop river basin management plans to include assessments of human activities and other impacts with economic assessments of river value (CEC, 2000). Additional EU Directives include: the EU Habitats Directive for the conservation of threatened flora and fauna as well the habitat considered vital to their conservation and as such an important driver of river rehabilitation; the Surface Water Protection against Pollution (2006/397/EC) in support of the WFD to set standards for concentrations of 41 priority chemical substances in surface water bodies that are known to cause risk to aquatic ecosystems (loss of habitat and biodiversity) and human consumption (CEC, 2006a); the Freshwater Fish Directive

(2006/44/EC) that protects and/or aims to improves the physical and chemical water quality of those freshwater bodies fit for sustaining salmonid and cyprinid populations (CEC, 20006b); and the Groundwater Daughter Directive to the WFD (2006/118/EC) for the prevention and regulation of groundwater pollution (CEC, 2006c). These Directives however only apply to certain designated water bodies as set by European Commission member states based on a set of ecological health criteria.

### 1.6.2 From local-scale form-led river rehabilitation to a catchment-level process-driven approach

River rehabilitation towards the end of the 20th century was based primarily on reach-scale single-species oriented designs led by single interest river user groups (Holmes, 1998; Harper et al., 1999; Kondolf et al., 2007). Angling groups and fishery managers were significant drivers with mostly salmonid based interests and a strong focus on physical habitat improvement schemes (White and Brynildson, 1967; Boon, 1998; Mainstone and Holmes, 2010). The fundamental theory underpinning this form-led river rehabilitation assumed habitat heterogeneity was the control mechanism for biological recovery (see White and Brynildson, 1967; Finnigan et al., 1980; Kondolf and Micheli, 1995; Bond and Lake, 2003). Installation of habitat structures such as flow deflectors, gravel riffles, large woody debris and bank covers were considered sufficient for ecological recovery. The form-led approach, however, does little to address the underlying cause of degradation through ignorance of catchment control variables. It inadequately considers hydrogeomorphic process (Sear, 1994; Boon, 1998; Clarke et al., 2003) and underlying environmental issues that persist at a much larger scale (Bond and Lake, 2003; Palmer et al., 2005). Kondolf and Micheli (1995) report that ecological recovery through enhancement of physical complexity alone had very poor levels of success and suffered from failure in the long-term. Post rehabilitation studies using macroinvertebrates (Harrison et al., 2004; Palmer et al., 2010) and fish (Pretty et al., 2003; Stewart et al., 2007) as indicators of ecological recovery concluded that habitat heterogeneity was not a determinant of increased biodiversity and it should not be the primary rehabilitation approach.

Many underlying environmental constraints act in combination, frequently at the larger catchment scale (urbanisation, land-use, diffuse pollution, abstraction), to impair biodiversity and population abundance. Although hydrogeomorphological processes are fundamentally associated with ecological functionality and biotic diversity (Harper et al., 1999; Bunte, 2004), water quality is increasingly being recognised as a primary limiting factor to biological recovery (Kondolf et al., 2007). However, addressing deficiencies in physical habitat remains vital to rehabilitation strategies. Approaches that incorporate increased structural complexity should attempt to provide for all the life history stages and allow natural recovery processes to dominate (de Jalón, 1995). A detailed knowledge of life-stage habitat characteristics as well as the structure of the population within the target river is therefore a prerequisite condition. Sear (1994) argues for a shift away from form-led schemes to greater emphasis on process-based geomorphology and an integrated catchment scale approach to river rehabilitation. A greater degree of sustainability is achieved at a larger scale (Sear, 1994; Bradshaw, 1996; Hey, 1996; Harrison et al., 2004; Beechie et al., 2012), where catchment level process-driven rehabilitation invests resources in site-specific factors found ecologically limiting (Palmer et al., 2010). These management plans must be delivered in combination with enhanced structural complexity (Pretty et al., 2003) through the reinstatement of hydrogeomorphological processes. Furthermore natural recovery has greater economic benefits (Sear, 1994; Bradshaw, 1996). The disadvantages of natural recovery, however, are the long temporal scales of recovery (Petts and Calow, 1996) dependent on the nature of stream power and sediment supply (Brookes, 1992).

### 1.6.3 Rehabilitation by means of spawning gravel introduction

Stocking a water body with *S. trutta*, generally in response to angling pressures, provides limited ecological recovery of a population, and in many cases dilutes wild *S. trutta* genetics. Rehabilitation of ecosystem structure by means of gravel introduction in an attempt to augment salmonid population recruitment through spawning provides a preferable alternative to a regulated stocking programme (de Jalón, 1995; Pedersen et al., 2009) and is an important rehabilitation tool (CALFED, 2005; Merz et al., 2006; Barlaup et al., 2008). Introduction of spawning gravel is a common rehabilitation technique (Jutila, 1992; Huusko and Yrjänä, 1997; Bunte, 2004; Merz and Setka, 2004; Singer and Dunn, 2006; Barlaup et al., 2008; Pulg, et al., 2013). Third sector organisations, such as the Wild Trout Trust and Rivers Trusts, as well as governmental agencies advocate the use of rehabilitation gravel (White and Brynildson, 1967; Finnigan et al., 1980; Wild Trout Trust, 2012). However there are few documented case studies and fewer still where success has been scientifically assessed, particularly true in the UK. Gravel rehabilitation for the augmentation of salmonid populations involves introducing a veneer of loose gravel within a known size range for spawning (see below) to suitable river reaches. As observed elsewhere (e.g. Kondolf and Wolman, 1993), the size of gravel used is important. Barlaup et al. (2008) found that a greater range of gravels had better success than a narrower distribution. A greater range of gravel size will support a wider size range of spawning fish (Kondolf and Wolman, 1993; Kondolf, 2000; Armstrong et al., 2003; Louhi et al., 2008).

It has been strongly argued that rehabilitation by means of gravel introduction should be used in conjunction with additional process-led designs based on a thorough understanding of the cause of ecological decline (Bunte, 2004) to increase longevity and potential sustainability given required hydraulic conditions (Wheaton et al., 2010). Rehabilitation gravel composition needs to be self-sustaining and naturally dynamic in order to provide long-term ecological benefits. In addition, reconstructing a riffle-pool morphology requires careful consideration of localised hydrogeomorphological control variables, such as two-dimensional modelling of shear stresses. In this respect Petersen et al. (1992) argued that gravel should only be introduced to streams with a suitable gradient and substrate present or the system should be allowed to recover through a process-led rehabilitation approach. The upper size limit of expected female *S. trutta* length should determine the area ($m^2$) of the installed gravel habitat. For example it is well known that redd size (the nest area where *S. trutta* eggs are deposited) correlates positively with fish size, approximately 3.5 times the length of fish (Crisp and Carling, 1989; Barlaup et al., 2008).

Gravel can be introduced to a river in two ways for rehabilitation purposes; directly into the channel location where habitat requires improvement, or as a bulk introduction at an upstream location that relies on natural stream processes to distribute gravels downstream to form semi-natural deposits (Bunte, 2004; Wheaton et al., 2004). Studies have shown that direct introduction yields immediate short-term physical improvements but greater susceptibility to redistribution by high flows and fine sediment deposition (Bunte, 2004). Furthermore, this technique has greater risk of accruing additional maintenance costs. Each method encourages salmonid spawning by creating a habitat with suitable parameters for embryo incubation; improved water velocity over shallow gravel (Huusko and Yrjänå, 1997) supports good interstitial permeability and thus delivery of dissolved oxygen to developing embryos. In this manner rehabilitation gravel also provide greater habitat variability suitable for many other species and can therefore facilitate an increase in ecological structure and complexity (Gore et al., 1998; Poudevigne et al., 2002; Gore et al., 2003; Barlaup et al., 2008).

The addition of gravel helps regulate inter- and intraspecific competition for vital life-stage specific habitat (Heggenes et al., 1999; Armstrong and Griffiths, 2001). Ideally rehabilitation gravel should include large cobbles spaced intermittently on the surface and along marginal areas to provide rearing and nursery habitat for juvenile fish (Jutila, 1992). Bunte (2004) contends that gravel rehabilitation schemes that do not make provision for juvenile life-stages will have little success. Good fry rearing habitat is a vital component of population recruitment and is required to be within close vicinity of spawning gravels so that emerging fry have less distance to travel and as such less prone to fatigue and predation (Pedersen et al., 2009).

Pasternack et al. (2004) modelled alternative rehabilitation spawning gravel designs under a range of flows in the Mokelumne River, California (Figure 1.6). Flat bank to bank gravel designs (gravel installed uniformly over a site) with intermittent surface boulder placements produced the best habitat quality for spawning and susceptibility to erosion for a given velocity but not for a range of velocities. A bar and braided design produced high habitat quality under a wider range of flows (Pasternack et al., 2004).

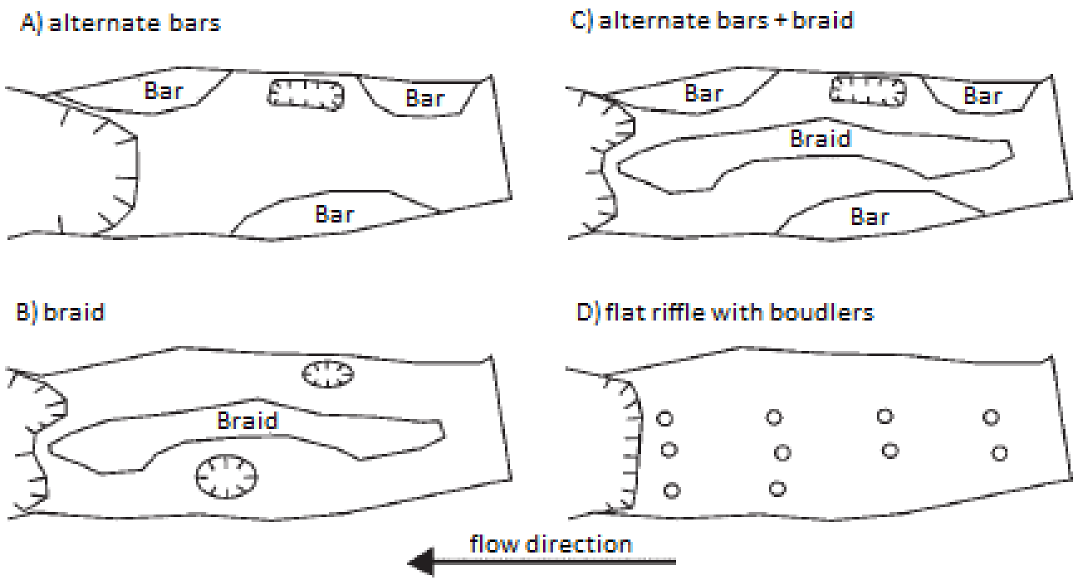



**Figure 1.6** Four alternative rehabilitation gravel designs. Hatched lines indicate pools, and the circles in diagram D) are boulders (from Pasternack et al., 2004).

### 1.6.4 Longevity of rehabilitation gravel and effects on *S. trutta* population augmentation

Although the short-term benefits of gravel augmentation schemes have been well demonstrated (Merz et al., 2004; Merz and Setka, 2004; CALFED, 2005; Barlaup et al., 2008; Pedersen et al., 2009), questions remain regarding the longevity and potential for sustained *S. trutta* population recruitment. Downstream loss of gravels due to scour and elevated flood velocities are common following gravel introductions (White and Brynildson, 1967; Merz et al., 2006; Barlaup et al., 2008), particularly where no supplementary rehabilitation has taken place (Pedersen et al., 2009). For example, Merz et al. (2006) observed up to 50% displacement of gravel in the Mokelumne river, California, over a four year period post gravel rehabilitation. Although the remaining gravels may retain some functionality, successful spawning generally decreases with gravel loss (Pedersen et al., 2009). Excessive sedimentation is another significant problem that threatens projects of this nature. Fine sediment, frequently derived from catchment soil erosion, smothers gravel habitat decreasing physical complexity and the ability of fish to spawn (Merz et al., 2006).

While gravel augmentation projects are common, pre- and post-monitoring and evaluation assessments are limited (Kondolf and Micheli, 1995; Kondolf et al., 2004; Wheaton et al., 2004). The installation and design of gravel-bed rehabilitation schemes are frequently based on local knowledge as opposed to rigorous scientific investigation, increasing the potential for medium- to long-term failure (Wheaton et al., 2004). Additionally, there are very few published studies (see Pasternack et al., 2004, Barlaup et al., 2008; Pedersen et al., 2009) or accessible unpublished reports that investigate how well artificially introduced gravel habitats function over the medium- to long-term (Kondolf and Micheli, 1995; Merz and Setka, 2004). Those studies that invest in post-rehabilitation monitoring define project success over the short-term with respect to the occurrence of redds observed on introduced gravels and/or the quantification of fry emergence density; largely a function of gravel availability (Barlaup et al., 2008; Pedersen et al., 2009). Although such studies provide an indication of potential short-term population recruitment, they fail to account for factors controlling mortality, either during the intragravel phase or at post emergence, and thus fail to evaluate long-term ecological success. Merz et al., (2004) provided a rare study of the physical parameters that govern egg mortality rates in rehabilitation spawning gravels in the Mokelumne River, California. In this study a rigorous scientific assessment based on egg-to-fry (ETF) survival rates of rehabilitation gravel were assessed in relation to sediment grain-size distributions, dissolved oxygen, temperature, and interstitial permeability. Results indicated that rehabilitation gravel reflected natural sedimentary characteristics associated with the Mokelumne River, and illustrated a mean 25% increased embryo survival for up to a 5 year period post-installation. However, the authors concluded that benefits of rehabilitation gravel in the Mokelumne River required sustained sediment mobilisation flows to maintain the viability of spawning habitat.

Ecological rehabilitation is frequently not attained over the short-term and at least a decade of post-rehabilitation monitoring is required to determine whether success can be feasibly achieved over the desired time scales (Kondolf and Micheli, 1995; Bradshaw, 1996). Natural rates of recovery tend to persist over the long-term and should be monitored accordingly. Gravel augmentation projects have a significant effect on sediment storage and flux rates, dependent on gravel size, sorting coefficients and local hydrogeomorphic processes (Singer and Dunn, 2006). For these reasons, the design of any rehabilitation spawning gravel project should thoroughly consider hydrogeomorphic processes at the catchment scale to provide the greatest enhancement potential and success over the long-term (Iversen et al., 1993; Kondolf and Micheli, 1995; Kondolf et al., 2004; Wheaton et al., 2004).

### 1.6.5 Case Study: Harpers Brook, Northamptonshire

Harper's Brook, an extensively channelised lowland tributary stream of the River Nene running through Northamptonshire, had 27 rehabilitation gravel structures 7-8 m in length installed in 1992 to support biological recovery. In 1995 25% of the installations no longer functioned as designed due either to excessive fine sediment accumulation within gravel interstices or a loss of critical stream velocity due to the natural compaction of gravels (Harper et al., 1998). Given the minimum 10 year monitoring period stressed by Kondolf and Micheli (1995) further design failures can reasonably be expected and the cost-benefit against annual maintenance is no longer an attractive proposition. Harper's Brook rehabilitation was based on a form-led design of isolated stream reaches in which hydrgeomorphology was used only for locating positions and proportions of the riffle-pool morphology whilst larger scale catchment regulated processes were not accounted for.

### 1.7 Study aims and objectives

Rigorous scientific analysis of the efficacy of gravel artificially introduced into stream channels for salmonid spawning and population production are sparse, particularly in light of the frequency that gravels are added to the streambed substrate (see Stewart et al., 2007). This gap limits our understanding of sustainable salmonid recruitment management.

The recent introduction of rehabilitation gravel to the River Stiffkey in North Norfolk provides an opportunity for quantitative and qualitative analysis at variable scales from catchment processes to microhabitat function. It is expected that the rehabilitation gravel will evolve and change over the short- to medium-term through interrelated physical processes. Key challenges include the characterisation of spawning gravel substrate at various spatial and temporal scales that define the habitat and its wider physical context; an assessment of change in these habitats that alter in response to hydrogeomorphic processes, and the effects on habitat quality and, in particular, how this relates to embryo development and recruitment at the population level. It is essential therefore that the streambed be considered a 4-D hydrogeomorphic unit, which varies laterally, horizontally and vertically over time.

This study evaluates the effectiveness of rehabilitation gravel as a river management tool through developing a greater understanding of the biological function of rehabilitation gravel environments in relation to physical catchment scale processes. The central aim of this study is to better understand the role of rehabilitation gravel with respect to the reproduction and recruitment of *S. trutta* populations. This research is underpinned by four key research objectives that apply to the River Stiffkey in North Norfolk:

- **Characterise the hydrogeomorphological context of the River Stiffkey**. Catchment scale hydrogeomorphic processes determine the sustainability of form-led habitat rehabilitation. This objective identifies catchment controls and hydrogeomorphic processes that define the physical character of the river. The causes of excessive sedimentation are determined.

- **Evaluate the physical characteristics of the rehabilitation spawning gravel habitat**. This objective will determine the physical suitability and the morphosedimentary nature of rehabilitation gravel as a spawning habitat.

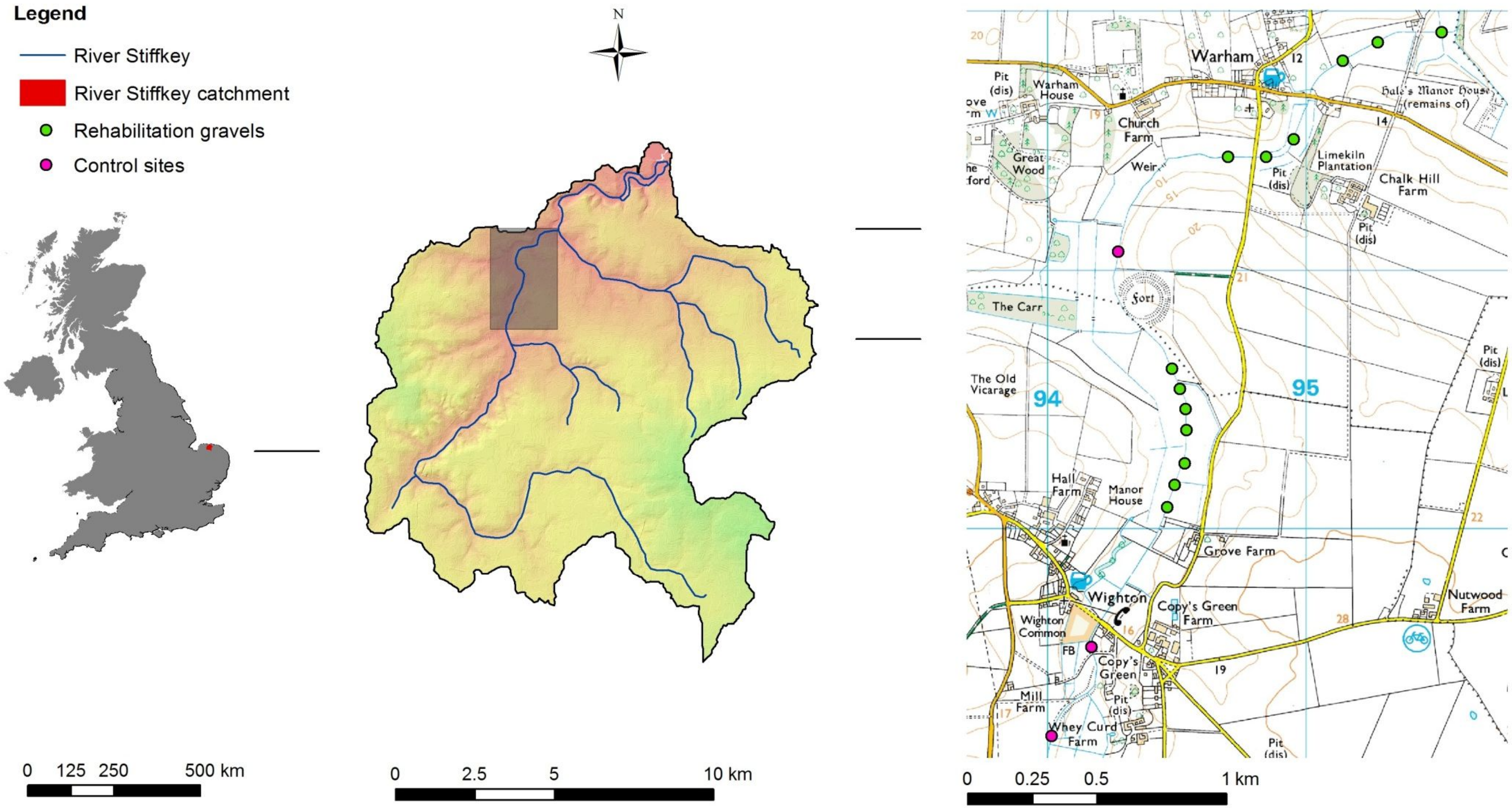
- **Investigate the influence of rehabilitation gravel on *S. trutta* population recruitment at the embryo stage of the life cycle**. This evaluation will estimate embryo survival for rehabilitation gravel and ascertain physical constraints that limit recruitment at this stage.

- **Determine the potential juvenile production capacity of the River Stiffkey**. This objective examines spatial relationships between key life-stage dependent habitat to identify an appropriate scale of stream management based on natural juvenile habitat abundance. In this manner key limitations to *S. trutta* production are identified.

### 1.8 Study site

This study focuses on a small chalk stream in North Norfolk, eastern England, the River Stiffkey (Figure 1.7). Rising at Swanton Novers, the river flows in a south to north direction, through the villages of Great Snoring, Walsingham, Warham, Wighton and Stiffkey into the Blakeney Channel and discharges into the North Sea through the Stiffkey saltmarshes. Along with its most notable tributary, the Binham Stream, the Stiffkey drains a catchment area of approximately 140 $km^2$ through its 30 km length, the largest catchment in North Norfolk (Environment Agency, 2005; Holt-Wilson, 2014). The large drainage basin relative to a short, low order stream is characteristic of many English chalk streams (Berrie, 1992). Most (76%) of the River Stiffkey channel baseflow is groundwater dominated, originating from the underlying Cretaceous Chalk dominated bedrock (Figure 1.2). Flow in the upper reaches is dominated by run-off from the tills of the Sheringham Cliffs Formation as well as the glacial sand and gravel deposits of the Briton's Lane Formation (Hiscock et al., 1996; Pawley et al., 2008; Environment Agency, 2013). Channel flow downstream of Thorpland Hall is however groundwater dominated and the lower 23 km of the river are classified as a chalkstream (Holt-Wilson, 2014).

The Stiffkey catchment receives a low annual average rainfall of approximately 570-670 mm (Environment Agency, 2005) with aquifer recharge occurring during winter (Figure 1.8, Table 1.2). The river is designated as over-abstracted (a deficit of 9.7 Ml $d^{-1}$ during low flow conditions), and is the only river in the north Norfolk Catchment Abstraction Management Strategy (CAMS) with this assessment status (Environment Agency, 2005; Environment Agency, 2008). The catchment is subject to sporadic but intense convective rainfall events during the summer months. The resulting run-off erodes and deposits large quantities of agricultural sediment into the river channel. Both the Chalk and Crag aquifers are key components of North Norfolk river ecosystems. Extensive Middle Pleistocene glacial deposits of shallow marine origins that overlay the chalk aquifer alter the characteristic chalk stream hydrograph response to precipitation (Hiscock et al., 1996; Pawley et al., 2008; Environment Agency, 2013). A shallow mud deposit, the Palaeogene Clays (London Clays), separates the underlying chalk bedrock from the overlying Crag which inhibits recharge of the chalk aquifer and increases stream flow response to precipitation (Ander et al., 2006).

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British Isles data from DIVA GIS (http://www.diva.org/Data)

**Figure 1.7** The study site, extending from the confluence with the Binham Stream to approximately 1 km upstream of Wighton village bridge, includes all of the introduced rehabilitation sites. This is an area of relatively low gradient. Exposed chalk is evident along the gravel-bed.

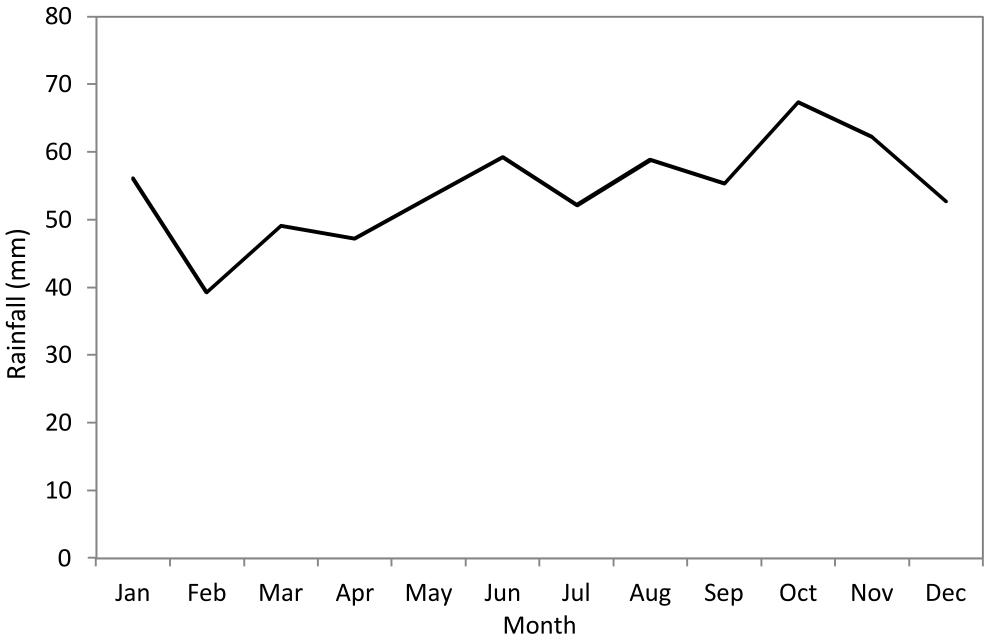



**Figure 1.8** 1981-2010 average rainfall (mm) for Marham, Norfolk, approximately 40 km from site. Peak rainfall occurs during the early winter months October and November**.** Data were obtained from the *UKCP09 The climate of the UK and recent trends* report (Jenkins et al., 2009).

**Table 1.2** Summary table of gauged daily flow ($m^3$ $s^{-1}$) recorded at Warham EA gauging station (TF9443241388), 1972-2013**.**

| Gauged daily flow ($m^3$ $s^{-1}$) | | SD |
|---|---|---|
| Low | 0 | - |
| High | 12.8 | - |
| Mean | 0.58 | 0.56 |
| Spring mean | 0.64 | 0.45 |
| Summer mean | 0.42 | 0.58 |
| Autumn mean | 0.50 | 0.62 |
| Winter mean | 0.78 | 0.51 |

Based on Environment Agency (EA) General Quality Assessment (GQA), a measure of river health based on a macroinvertebrate monitoring survey, water quality in the River Stiffkey ranges from good to very good (Environment Agency, 2005). The river receives a consented Sewer Treatment Works (STW) discharge (Environment Agency, 2005). Water quality suffers particularly from nitrate enrichment from point and diffuse sources derived from the agricultural dominated catchment (Ander et al., 2006; Environment Agency, 2013). A report by

Natural England (2013) observed that the River Stiffkey was the most at risk of diffuse pollution in North Norfolk based on the numerous sources of contamination within its catchment. Additionally, an excessive agricultural derived sediment load has reduced the quality of much of the remaining spawning habitat.

Extensive stretches of the River Stiffkey have been subjected to flood regulation between Wighton and Warham during the 1970's and 1980's (Pawson, 2008). Consequently these areas are over-widened and/or over-deepened with concomitant impacts on stream ecology and a significant loss of salmonid spawning habitat. This is most apparent below Wighton where the channel is both over deepened and widened. There are several UK BAP species present in the River Stiffkey: European otter (*Lutra lutra*), water vole (*Arvicola terrestris*), brook lamprey (*Lampeta planeri*) and since its re-introduction in 2011, native white-clawed crayfish (*Austropotamobius pallipes*) (UK Steering Group, 1995b; Environment Agency, 2005). The lower reaches of the river, the Stiffkey Valley, are a designated SSSI as this area has significant wetland habitat and bird biodiversity, including populations of breeding avocet (*Recurvirostra avosetta*) (Oddy, 2014). Fish species diversity is average for North Norfolk rivers and includes the European eel (*Anguilla anguilla*), brown trout (*S. trutta*), 3-spined stickleback (*Gasterosteus aculeatus*), 9-spined stickleback (*Pungitius pungitius)*, stone loach (*Barbatula barbatula*), gudgeon (*Gobio gobio*), European bullhead (*Cottus gobio*), brook lamprey (*Lamperta planeri*) and flounder (Pawson, 2008). The river has a self-sustaining population of *S. trutta* throughout its length (Environment Agency, 2005; Pawson, 2008). Migratory *S. trutta* (sea-trout) are known to enter the river and there are anecdotal accounts of a few fish caught by the Holkham Angling club each year (Pawson, 2008). Additionally, several migratory *S. trutta* were caught in electrofishing surveys approximately 1.5 km downstream of Warham in 2012 (Wright, pers. comm., 2012).

Although resident *S. trutta* are present in all North Norfolk rivers, it is estimated that just 5% of total river length is accessible to migratory *S. trutta* (Pawson, 2008). Sea sluices in Cley, mill structures at Glandford, Letheringsett and Thornage are all barriers on the River Glaven, and the Burnham Mill prevents access to spawning reaches upstream on the River Burn (Pawson, 2008). The River Stiffkey has no migration barriers since the sea sluice gate was modified to a more fish friendly flap under the Living North Sea project by the Environment Agency in 2009. Both the Rivers Stiffkey and Glaven have confirmed catches of migratory *S. trutta* in the lower reaches although suitable spawning habitat is limited in the region (Pawson, 2008).

The River Stiffkey was considered a good candidate for rehabilitation gravel as there are no migration barriers preventing fish from accessing upstream reaches (Pawson, 2008). Three spawning gravel habitats were installed in 2003 as part of a Wild Trout Trust (WTT) project. In 2009 a further 10 rehabilitation gravel habitats were installed as part of the Anglian Rivers Sea Trout Project (ARSTP) and Living North Sea (LNS) project (Figure 1.7). Rehabilitation gravels were of the flat bank to bank gravel design, similar to those modelled by Pasternack et al. (2004), however without intermittent surface boulders. The LNS, an Interreg IVB funded programme (concluded in 2012), aimed to achieve improvements in sustainable management of North Sea fish stocks as well as address limitations to population recruitment in the freshwater phase of fish life cycles. Further aims were to reduce deficiencies in the freshwater spawning phase of migratory *S. trutta* by providing a favourable environment for embryo incubation and fry rearing. At the local level a key factor that limits production of *S. trutta* in Anglian Rivers is the loss of spawning habitat. Land-use changes within the River Stiffkey catchment have led to increased fine sediment loads (Pawson, 2008). Further, the suspension and deposition of these sediments have deteriorated the quality of remaining salmonid spawning habitat. The ARSTP received EU Interreg funding from the LNS project to conduct habitat improvements on the River Stiffkey. The habitat rehabilitation programme involved introducing spawning gravel into river sections where this was lacking chiefly due to channel deepening and widening for flood prevention purposes (Gill et al., 2009).

Following the installation of the rehabilitation gravel under the Living North Sea project, the Norfolk Rivers Trust aims to rehabilitate the biodiversity of the River Stiffkey through linking several key landowners and targeting water quality and habitat deficiencies at a catchment level approach (Norfolk Rivers Trust, 2014).

# 2 Scientific approach

## 2.1 Research focus and problem statement

Salmonid populations have declined worldwide largely through the loss and degradation of spawning and rearing habitat associated with river channel modification and catchment land-use (see Petrosky et al., 2001; Borsuk et al., 2006; Walter, 2015). A key objective of the Living North Sea (LNS) project was to redress habitat deficiencies required for the freshwater phase of the migratory *S. trutta* (sea trout) life cycle. Introduction of rehabilitation gravel to the mid-reaches of the River Stiffkey were intended to encourage accessibility of migratory *S. trutta* and to augment non-migratory *S. trutta* populations, and indirectly encourage their seaward migration through increased population-density factors, such as competition and predation. The River Stiffkey has however been subject to excessive sediment load pressures associated with river engineering and poor land management. Furthermore, chalk streams have low stream power and a limited capacity to prevent in-channel sediment deposition. Rehabilitation gravel longevity is therefore precarious due to the susceptibility of fine sediment deposition.

This research focuses on the suitability of rehabilitation gravel for *S. trutta* population recruitment in the River Stiffkey. It combines hydrogeomorphology, geography and biology through a hierarchy of relevant spatial scales; catchment level land-use and geomorphic processes drive sedimentary parameters fundamental to embryo survival at the microhabitat scale, and spatial relationships between juvenile key life-stage habitat types at the reach scale that are responsible for regulating population production.

## 2.2 Research design

This research is a case study that investigates sedimentary characteristics and spawning quality of rehabilitation gravel introduced into the River Stiffkey. Three rehabilitation gravel structures were installed in 2003 and a further 10 in 2009 forming 13 flat crested bank-to-bank gravel deposits. These were arranged in 5 separate pool-riffle sequences over a 3 km stretch of river (see Chapter 1, Figure 1.7). Given that these rehabilitation gravel structures were installed to similar sedimentary specifications, flint rejects and small boulders (100-174 mm) overlain by gravels (10-40 mm) (T. Jacklin, pers. comm., 17/01/2011) but separated by 6 years of exposure to fluvial processes, they were considered as two treatments: gravels installed in 2003 (referred to hereafter as sites 2003A-C) and gravels installed in 2009 (referred to hereafter as

2009A-J) (Figure 2.1). A third treatment of naturally occurring spawning gravels was incorporated to control for sedimentation processes that physically distinguish between the two rehabilitation gravel treatments. Three sites (referred to hereafter as Whey Curd, Water Hall and Fort) were selected based on known salmonid spawning preference parameters; 15-10 cm water depth with a velocity between 0.20-0.75 m $s^{-1}$ in an area with abundant substrate within the $64>D\geq16$ mm range (Jutila, 1992; Kondolf and Wolman, 1993; Armstrong et al., 2003; Louhi et al., 2008; Marchildon et al., 2010). Natural spawning gravel suitable for freeze core sampling was scarce within the 3 km covered by rehabilitation gravel. Two control sites, Water Hall and Whey Curd, were therefore located further upstream.

Catchment processes and historical land-use data, particularly those forces driving current hydrological and sedimentological processes, contextualise catchment controls that characterise the sedimentological composition of rehabilitation gravel.

Form-led rehabilitation designs, an approach based primarily on increasing structural complexity, fail to recognise the importance of catchment level environmental constraints to ecological recovery. Greater appreciation of these constraints supports more suitable site-specific rehabilitation strategies with superior built-in sustainability and reduced potential for maintenance.

An examination of the sediment composition of rehabilitation gravel is used to characterise gravel quality and suitability for spawning. Differences in grain-size distributions between rehabilitation gravels installed in 2003 and those installed in 2009 provide an indication of habitat longevity. A survey of natural spawning gravel abundance throughout the study site determines the naturally available grain-sizes used for spawning. Comparisons with rehabilitation gravel provide an indication of their role in *S. trutta* population recruitment.

Egg-to-fry (EFT) survival derived from in-stream egg-box experiments provides estimates of embryo survival within rehabilitation gravel. EFT survival is examined in association with incubation sediment composition to determine detrimental effects of fine grained sediment. Embryo survival examined in this manner provides a further indication of habitat quality and potential for population recruitment in the River Stiffkey. It also illustrates the biological longevity in association with the morphosedimentary character of rehabilitation gravels.

Post-emergent fry illustrate a maximum migration distance between life-stage dependent habitat types, beyond which mortality significantly increases. Determination of the spatial relationship between these habitat types is used to identify key areas of *S. trutta* production within the study reach. As migration commences from spawning habitat, spatial proximity from rehabilitation gravels to the next key life-stage habitat (nursery habitat) is vital to *S. trutta* recruitment in the River Stiffkey. Fragmented spatial relationships between juvenile habitat regulate *S. trutta* populations and as such investigation at this scale is particularly suitable for the development of rehabilitation strategies aimed at population recruitment.

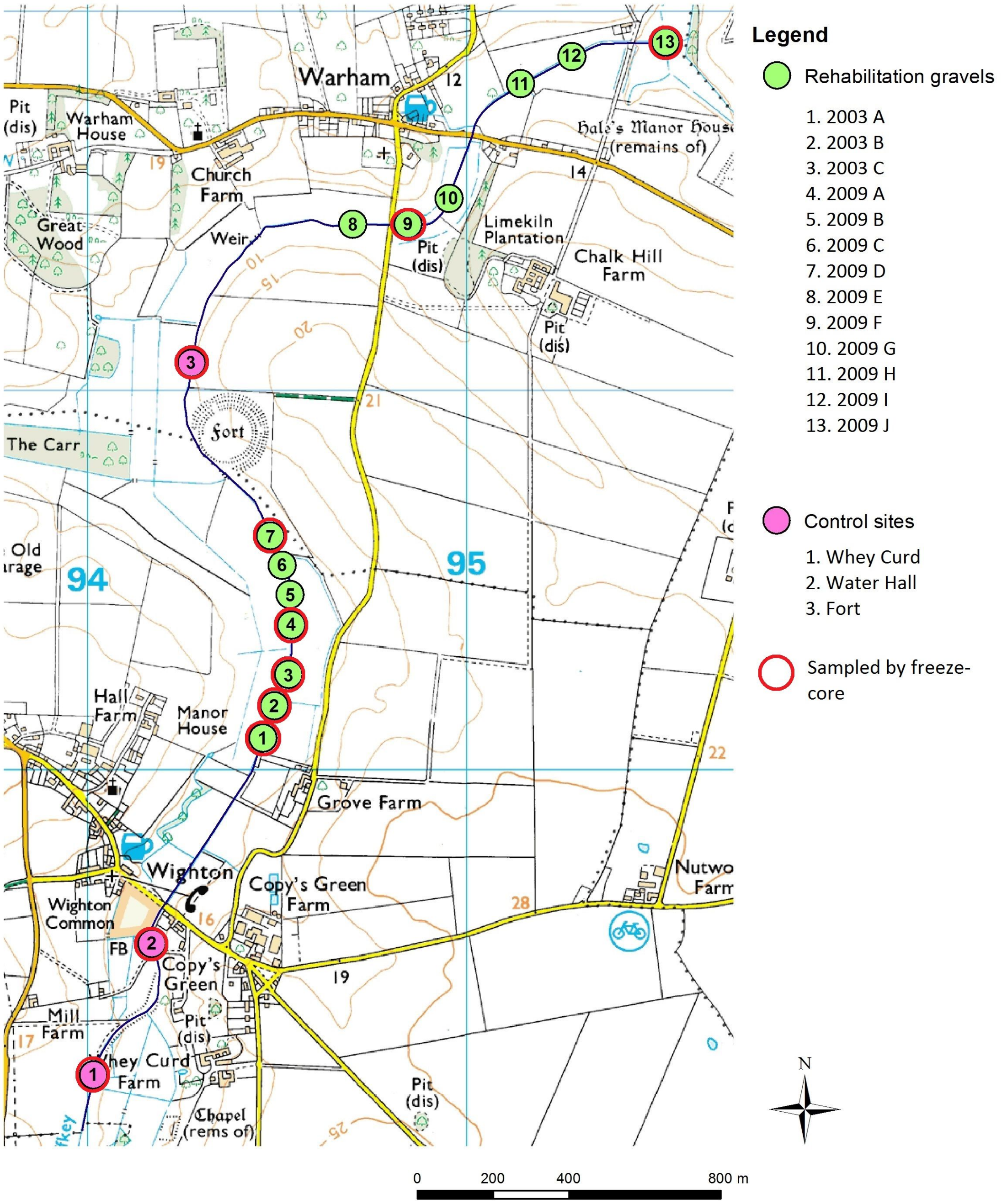





**Figure 2.1** Location of rehabilitation gravel and control sites. Note the single control site (3: Fort) located between rehabilitation gravel sites 2009D and 2009E. This reflects the scarcity of natural gravels suitable for S. trutta spawning downstream of village. Site sampled by means of freeze-core are highlighted.

## 2.3 Methodology

### 2.3.1 Catchment controls on river channel hydrogeomorphology

This investigation was based on a desktop study. Initially the River Stiffkey catchment area was defined and key flow pathways derived. The catchment area was delineated from a 5x5m Digital Terrain Model (DTM) using the ArcHydro toolkit (v10.2) in ArcMap (v10.2). Sinks were filled to remove small imperfections, or areas of no data, from the DTM. Flow direction was calculated using the D8-flow algorithm to determine the direction of flow out of each cell in the DTM dataset. The accumulated flow into each cell, calculated by weight of the flow direction, defined stream channel pathways. The upstream catchment area that contributed flow to a defined common point, the pour point, was defined. Key catchment flow pathways were derived from the flow accumulation data by extracting cells with a catchment of ≥0.025 $km^2$ (i.e. cells with at least 10,000 cells (5x5 m) flowing into them) based on ground truthing of the actual river channel. Historic and contemporary catchment controls and channel hydrogeomorphology were examined to identify changes that could have resulted in excess sediment delivery to the river channel. The following data sets were used:

- Bedrock and superficial geology of North Norfolk

  Superficial and bedrock geology data mapped at 1:50 000 scale were obtained from the Digital Geological Map of Great Britain (DiGMapGB-50) dataset (Smith, 2013). Superficial deposits consist of natural unconsolidated surface material (drifts) of Quaternary age (< 2.6 million years). Superficial deposits rest on the solid consolidated bedrock geology of pre-Quaternary age. Data were analysed in ArcMap (v10.2).

- Topography of North Norfolk

  Ordnance Survey 5x5 m Digital Terrain Model (DTM) data were used for topographic mapping. The Hillshade tool in ArcMap (v10.2) was used for visualisation of elevation throughout the River Stiffkey catchment. 10m contour lines were superimposed from an Ordnance Survey 5x5 m Contour dataset.

- River Stiffkey palaeo-channel analysis

  A special non-commercial licence for Light Detecting and Ranging (LiDAR) Composite data (November 2013) was obtained from the Environment Agency Geomatic Archive Data Team for a 2 m resolution LiDAR Digital Terrain Model dataset. LiDAR data were used to determine the location and physical characteristics of palaeo-channels within the catchment. Palaeo-channels are evident by altering the minimum and maximum altitude range of the data whilst systematically exploring 50-100 m of river channel from headwaters to outlet.

- Long-term monthly-averaged precipitation

  Long-term precipitation data 1910-2011 was obtained from those data sets created for use in the UK Climate Projections (UKCP09) available from the Met Office and used in the *UKCP09 The climate of the UK and recent trends* report (Jenkins et al., 2009). These monthly-averaged data cover the UK with a spatial resolution of 5x5 km. Initial analyses were conducted within a GIS created in ArcMap (v10.2). Precipitation data for the delineated River Stiffkey catchment were isolated. Seasonal variation was investigated based on the meteorological calendar; spring (March-May), Summer (June-August), Autumn (September-November) and winter (December-February) was then investigated.

- Warham gauging station: discharge

  River Stiffkey discharge data from the Warham gauging station (station ID 24018) was obtained from the National River Flow Archive (NRFA). Monthly mean daily gauged flow data ($m^3$ $s^{-1}$) from 1 April 1972 to 30 September 2013 were examined.

- River Stiffkey catchment land-use

  Catchment land-use data were derived from historic and contemporary map data. 1849-1899 Country 1:10560 Series historic maps, available from the original mapping sheets which were scanned, georeferenced and divided up into National Grid Tiles based on contemporary Ordnance Survey maps from the Landmark Information Group and available from Edina Digimap services. The mapped river channel was traced in ArcMap (v10.2) to identify channel planform as a comparison against contemporary flow paths. Maps dating 1849-1899 and 1922-1969 were extracted based on the delineated catchment polygon. Land Cover Map of Great Britain (1990) (Fuller et al., 1994), Land Cover Map 2000 (Fuller et al., 2002) and the Land Cover Map 2007 (Morton et al., 2011) illustrated more recent land-use. Land-use classes were developed from summer-winter composites derived from satellite imagery. Analyses were conducted in ArcMap (v10.2).

Land Cover Map (LCM) of Great Britain (1990) consisted of a digital dataset derived from satellite imagery from the Landsat 5 Thematic Mapper. This dataset contained 25 land-use classes at a 25 m resolution, of which the following where used in analysis of the River Stiffkey catchment: Saltmarsh (intertidal sand, silt or mud habitats with halophytic grasses and herbs mapped up to normal high water spring tides), Beach and Coastal Bare (intertidal mud, silt, sand, shingle, and rocks), Bracken (herbaceous vegetation dominated by *Pteridium aquilinum*), Coniferous Woodland (coniferous species), Deciduous Woodland (bare in winter), Continuous Urban (large areas of development that completely fill each pixel to exclude permanent vegetation), Suburban/Rural Development (small areas of developed land that do not fill each pixel to include permanent vegetation), Dense Shrub Heath (plant communities with high content of heather on sandy soils), Grass Heath (coastal dunes and inland grasslands growing on sandy soils typically of an acidic nature), Inland Bare Ground (natural surfaces such as rock, soil, sand, gravel, including land denuded by cattle), Inland water (all map-able fresh waters), Meadow/Verge/Semi-natural (managed grasslands at a lesser extent than the Mown/Grazed Turf class), Mown/Grazed Turf (managed turf grasslands for either agricultural or amenity purposes), Rough/Marsh Grass (lowland herbaceous vegetation typically fens, marshes, saltmarshes, and derelict ground), Scrub/Orchard (deciduous vegetation typically including hawthorn and brambles), Tilled Land (arable cropland - received annual tillage, typically cereals), Unclassified (not allocated to any of the 25 land-use classes).

Land Cover Map (LCM) 2000 data were derived from a collection of satellite images to produce 221 land-use classes based on the UK Biodiversity Action Plan (BAP) at a 25 m resolution. Those classes used for characterisation of the River Stiffkey catchment included: Arable Cereals (arable cropland), Arable Horticulture (annual and perennial crops, including ploughed land), Broad-leaved/Mixed Woodland (>5 m high tree cover and scrub <5 m with >30% cover), Calcareous Grassland (of known pH >5.5), Coniferous Woodland (includes semi-natural stands and plantations, >20% cover), Dense Dwarf Shrub Heath (plant communities continuing high content of heather on sandy soils), Improved Grassland (distinguished from semi-natural grasslands), Inland Bare Ground (natural surfaces such as rock, soil, sand, gravel, including land denuded by cattle), Supra-littoral Sediment (sedimentary coasts defined by terrestrial mask such as beaches, mudflats, dunes and shingle beaches), Littoral Sediment (sedimentary coasts defined by maritime mask), Neutral Grassland (of known pH between >4.5 and <5.5), Saltmarsh (intertidal sand, silt or mud habitats with halophytic grasses and herbs mapped up to normal high water spring tides), Suburban/Urban Developed (small areas of developed land that do not fill each pixel to include permanent vegetation), Continuous Urban (large areas of development that completely fill each pixel to exclude permanent vegetation), Water (map-able inland fresh waters).

Land Cover Map (LCM) 2007 includes 23 land-use classes updating LCM 2000. Land-use classes included: Arable and Horticulture (annual and perennial crops, including ploughed land), Broadleaved Woodland (>5 m high tree cover with >20% cover, scrub <5 m with >30% cover), Coniferous Woodland (includes semi-natural stands and plantations, >20% cover), Heather Grassland (division of LCM 2000 Dwarf Shrub Heath separating out heather grasslands), Improved Grassland (greater productivity than semi-natural grasslands), Neutral Grassland (upgrades LCM 2000 grassland classes based on botanical composition), Rough Grassland (managed low productivity grasslands), Freshwater (open waters, canals, rivers and streams), Saltmarsh (intertidal sand, silt or mud habitats with halophytic grasses and herbs mapped up to normal high water spring tides), Littoral Sediment (sedimentary coasts defined by maritime a mask), Supra-littoral Sediment (sedimentary coasts defined by a terrestrial mask such as beaches, mudflats, dunes and shingle beaches), Urban (large areas of development that completely fill each pixel to exclude permanent vegetation), Suburban (small areas of developed land that do not fill each pixel to include permanent vegetation).

- Longitudinal profile of the river channel

The longitudinal profile of the river was surveyed using a dGPS (Leica System 1200) and Leica Viva GS15 rover unit. Mean positional accuracy was 1.4 cm. Data were post processed. Measurements were taken at 5 m intervals following a route directly down the middle of the stream channel except in places where plant density was prohibitive. Areas of excessive sediment accumulation made establishing the streambed difficult in places. These areas were measured at the point where the antenna axel pole sunk into unconsolidated sediment under its own weight.

### 2.3.2 Sedimentological analysis of rehabilitation gravels

Traditional volumetric sampling of fluvial gravels under flowing water conditions, such as the grab-sampling techniques, underestimated the percentage weight of fine material (Thoms, 1992). Freeze coring, which keeps fine material in-situ, provided a more representative sample with fewer samples (66% less) required than other techniques. A vertical grain-size distribution profile and substrate composition of spawning sediment sampled in this manner can be examined at specified depth intervals.

Nine freeze cores were extracted from all three sites in the natural and 2003 gravel treatments, and 12 cores were extracted from four 2009 gravel treatment sites (Figure 2.1). Substrata at each site were sampled in a non-random manner to ensure greater coverage; freeze cores were sampled at both the up- and downstream extent as well as from the mid-point of the site. A study into the efficacy of freeze cores as a suitable sample method concluded that 5 freeze core samples were necessary to characterise streambed substrata $D_{50}$ to within a 5% sampling error at 95% confidence levels (Thoms, 1992). Based on Thoms (1992) study, 3 freeze cores sampled at each site in this study does introduce >5% sampling error at 95% confidence levels for accurately determining spatial variability of population $D_{50}$ at the site scale of investigation. Church et al. (1987) established sample weight criteria for freeze core analysis based on the percentage composition of the coarsest particle investigated. These workers suggested that where the coarsest particle did not exceed 0.1% of total weight, a sample with a representative 64 mm grain-size should have a total weight of 362 kg (Figure 2.2). Milan et al. (1999) further refined the sample weight criteria established by Church et al. (1987) based on the influence of sediment sorting. Representative sampling of poor to very poorly sorted substrata were found to require greater total weight, between 782 to 1035 kg respectively (Milan et al., 1999).

Particle sorting was expected to differ between the gravel-dominated rehabilitation gravel and natural gravel sites. Natural gravel was expected to be poorly to very poorly sorted, whilst rehabilitation gravel was likely to be poorly to well sorted and as such not in need of the stringent sample weight criteria as suggested by Milan et al. (1999). It follows that a sample size of 362 kg was required to characterise a rehabilitation gravel treatment $D_{50}$ based on Church et al. (1987) criterion for 0.5% sampling error at a 95% confidence level for 0.1% total composition of 64 mm particle size. Mosely and Tindale (1985), however, reported more relaxed sample weight criteria as the largest particle should not constitute more than 5% of the total sample weight (Figure 2.2). Based on these criteria freeze core samples with 64 mm as the coarsest particle size require samples of 6 kg total weight. Such criteria had greater relevance for rehabilitation gravel characterisation as coarse particles constitute a greater composition of the grain-size distribution. Characterisation of a poorly to very poorly sorted natural gravel treatment $D_{50}$ under similar constraints required a sample size of between 782-1035 kg (Milan et al, 1999). However, the stringent sampling criteria as described by Milan at al. (1999) would require several hundred freeze cores from each natural gravel site in order to characterise spatial variability at this scale. Given the fragility of this precious resource in the River Stiffkey, such sample weight criteria were not considered ecologically viable and sample error is therefore implicit.

In this study individual freeze cores were analysed in order to describe spatial variability at the 5 cm scale within each site. However, core weights were combined to characterise individual sites and site weights were combined to describe gravel treatments, consistent with Wolcott and Church (1991). Based on these sample size weight criteria and a combined weight of freeze cores, sample error was successively reduced as the spatial scale of investigation was increased, from spatial variability described at a 5 cm scale for each site to gravel treatment characterisation.

Removal of frozen sediment from the core tube was a point of contention. The use of a chisel to dislodge frozen sediment from the core tube enabled rapid sampling of streambed substrata facilitating greater replicates over a sample period. However, there would be an unaccounted loss of sample accuracy. Coarse grains were likely to break into finer fractions and as such result in overrepresentation of the finer sediment fraction. Cross-contamination between stratification levels was also a concern and as such this method was not considered appropriate. An alternative and preferred method was to allow sediment to defrost and fall into stratification units under gravity. This method had reduced cross contamination, no overrepresentation of any grain-size and therefore had greater inherent accuracy. This method was however slower and as such fewer freeze cores replicates could be extracted over a comparable sample period.

Freeze cores were sampled using the freeze core method provided by the Centre for Environment, Freshwater and Aquaculture Science (Cefas), following the methods outlined in Milan (1996) and an adaption of the method described in Petts et al (1989) and Carling and Reader (1981). Migratory *S. trutta* excavate gravel and deposit their eggs up to a maximum of 30 cm depth (Crisp and Carling, 1989). Thus a 50 mm diameter and 1300 mm long stainless steel spiked-tube were driven 30 cm into the substrate. The core tube was closed and pointed at one end. An open-ended 50 $cm^2$ baffle was placed over the core tube and driven under hand-pressure approximately 5-10 cm into the surface spawning sediments diagonally across the river current (Figure 2.3a). This prevented interstitial gravel flow within the top sediment layers and allowed even freezing of the substrate. Approximately 10 litres of liquid nitrogen were steadily introduced to the core tube, freezing the associated substrate to the outside (Figure 2.3b). The amount of liquid nitrogen required varied with substrate composition; where sediments were less compacted more liquid nitrogen was required in order to freeze larger volumes of water within the interstitial spaces. The core tube and attached sample were winched out from the river bed using an A-frame and chain block and allowed to defrost over a stratification unit with 5 cm intervals (Figure 2.3c).

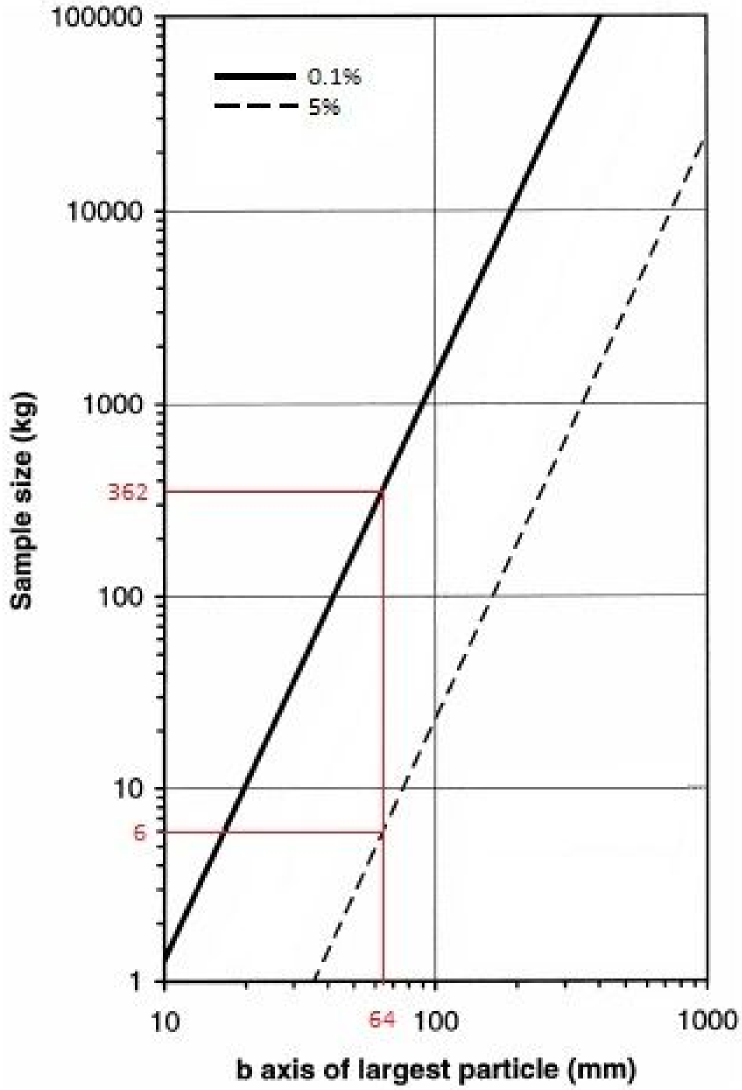



**Figure 2.2** Plot indicating bulk sample weight criteria required for representative population D50 estimation based on Church et al (1987), 0.1%, and Mosely and Tindale (1985), 5% (modified from Milan et al., 1999). The individual sample weight for a maximum particle of 64 mm at 0.1% and 5% is indicated.


**Figure 2.3** Gravel freeze-coring: core tube (funnel facilitates pouring of liquid nitrogen), baffle, and A-frame with chain block set-up (a), liquid nitrogen being introduced to the core tube (b), two cores defrosting on the stratification unit (c).

Each 5 cm gravel core increment was washed out of the sample bags into pre-weighed aluminium containers using distilled water. Samples were oven dried at a temperature of 40° C and weighed. These were then wet sieved at *D*<0.5 mm and the resultant fine sediment retained in solution was analysed in a laser granulometer. A single method to analyse complete grain-size distributions was desirable, however, compaction of the finer grained sediments <0.5 mm made this difficult. Samples were therefore divided by sieving at <0.5 mm to remove the smaller fraction and prevent errors in the sieving process in the coarser fraction.

The coarser fraction >0.5 mm was oven dried again, weighed and mechanically sieved for 10 minutes through sieve apertures of 60 mm, 30 mm, 16 mm, 8 mm, 6.35 mm, 5.66 mm, 4.76 mm, 4 mm, 3.35 mm, 2.83 mm, 2.38 mm, 2 mm, 1.68 mm, 1.4 mm, 1.2 mm, 1 mm, 0.841 mm, 0. 71 mm, 0.6 mm, 0.5 mm. The retained sediment fractions were weighed. Supernatant water was removed from the <0.5 mm sample and sediments were dried at 40° C and weighed. Samples <0.5 mm were moistened and mixed sufficiently well enough to ensure a heterogeneous sample. A sub-sample was analysed in a Malvern Hydro 2000 MU laser granulometer, capable of measuring at 99% accuracy within a particle size range of 2-0.0002 mm with less than 1% variation in reproducibility of samples (Malvern Instruments, 1997). The Malvern granulometer sizes particles based on the principle that light is diffracted off of a sediment grain at an angle inversely proportional to their size (Agrawal et al., 1991). Laser diffraction can be very precise. However, reported limitations include underestimation of the clay-sized particles compared to pipette and hydrometer methods (Buurman et al., 1996; Beuselinck et al., 1998; Esher et al., 2004, Blott and Pye, 2006) and overestimation of sand-sized sediment against sieving analysis (Hayton et al., 2001; Esher et al., 2004; Blott and Pye, 2006). Differences were attributable largely to particle sphericity. Grain-size distribution errors were smallest when particles approximated near perfect spheres (Jonasz, 1991; Blott and Pye, 2006). Such error may lead to overestimation of coarser size fraction based on cross-sectional area of non-spherical particles (Esher et al., 2004). Blott and Pye (2006) observed that software interpretation approximated log normality so very skewed and bimodal samples were poorly represented independent of particle shape. Samples are held in suspension and continually looped through a sample cell where they pass through a laser beam at a (user defined) speed of 2000 rpm. All sub-samples were measured for abundance of silica, which has a refractive index of 1.544. Sub-samples were suspended in tap water and dispersed by ultrasonic energy for two minutes. Those sub-samples observed to contain calcium carbonate ($CaCO_3$) were analysed for abundance of $CaCO_3$, which has a refractive index of 1.61. The mean of five runs of each sub-sample was recorded. The Malvern granulometer expressed the results of each size fraction as a percentage volume of the input sub-sample. A total weight (g) for each size fraction was calculated.

Stratigraphic plots of freeze cores illustrating the gravel-bed structure at 5 cm increments from the streambed surface to a depth of 30 cm were drawn using the *C2* software programme

(Juggins, 2003). The composition of gravel (64>$D$≥16 mm), coarse sand (2>$D$≥1 mm), clay particles ($D$<0.004 mm), finer grained sediment ($D$<1 mm) and median grain-size ($D_{50}$) were plotted. Clasts ≥64 mm were described based on axis dimensions (mm), with roundness based on the index of Powers (1953) and percentage weight contribution to grain-size distribution.

### 2.3.2.1 Spawning substrate quality indexes

Many researchers have reported, with varying results, on some qualitative threshold value of fine grained sediment that enabled a 50% alevin emergence rate from incubation substrate (Table 2.1). Based on the values outlined in Table 2.1, Milan et al. (2000) adopted a threshold value of 14% sediment <1 mm ($D$=0.83 mm) to determine the health of spawning gravel deposits. The same threshold value has been used for this research.

**Table 2.1** Gravel quality index illustrating the maximum percentage of sediment that enabled 50% emergence of salmonid embryos. Note the variability of figures. Modified from Kondolf (1988).

| **Source** | **Grain-size (mm)** | |
|---|---|---|
| | **0.83** | **2** |
| Cederholm and Salo (1979) | 7.5 | |
| Cederholm and Salo (1979) | 17 | |
| Hausle (1973) | | 10 |
| Hausle and coble (1984) | | 20 |
| Iwamoto et al (1978) | | 15 |
| Koski (1966) | 21 | |
| NCASI (1984) | 12 | |
| Reiser and White (1990) | 13 | |
| Tagart (1976, 1984) | 11 | |
| Mean | 13.6 | 15 |
| S.D. | 4.8 | 5 |

The sand index (SI) (Peterson and Metcalfe, 1981), a measure of the composition of sand in spawning substrate, was calculated as:

Equation 2.1:

$SI = Sc/16 + Sf/8$

where *Sc* is percentage weight of coarse sand ($0.5 \leq D < 2$ mm) and *Sf* is percentage weight of fine sand ($D < 0.5$ mm). A sand-index value smaller than 1 has been shown to be excellent for *S. trutta* emergence, whilst 1.5 is poor (Peterson and Metcalfe, 1981).

### 2.3.2.2 Identified constraints and errors of sediment analysis

Total exclusion of clasts ≥64 mm has been identified as a shortcoming of studies reporting on spawning substrate (Kondolf, 2000). Cobbles and small boulders ≥64 mm were extracted in the freeze core samples for this study. Due to their size these clasts spanned one or more 5 cm stratification boundaries and were therefore excluded from grain-size distribution analysis at the 5 cm scale. This placed an emphasis on those grain-sizes with a small weight contribution such as sediment <1 mm that were fundamental to the study as observed in Kondolf (2000). However, cobbles were included in statistical analysis at the core, site and treatment levels. Clasts ≥64 mm extending beyond the 30 cm core limit were analysed if the majority of the clast occurred in the upper 30 cm of the core.

Grain-size distribution analysis was not based on equal class limits. Sediments <0.5 mm were analysed in a laser granulometer that defined small but regular differences between particle sizes. Sediment >0.5 mm was analysed on a sieve stack with irregular class limits between sieve sizes. Some of this difference is inherent as smaller differences existed between smaller sediment particles, whilst small differences are less evident between larger clasts. This resulted in irregular class limits throughout the grain-size distribution; a greater amount of detail towards the fine grained tail of the distribution and less detail in the coarse grained tail. In order to achieve equal class limits between particle sizes an improbably large set of sieves incorporating all sizes would be required. Moreover, a size-range of sediment based on determined *S. trutta* spawning preferences were a key requirement for this study.

A net loss of sediment during the laboratory sieve analysis was observed. Cumulative weights of individual sediment size fractions rarely matched exactly the pre-analysed sediment sample weights. Sediment loss occurred as smaller sediment size classes, particularly sand $2 > D > 0.062$

mm, were transferred out of individual sieves for weighing. Sediment <0.004 mm (clay) adhered to the container sides during the initial drying process, which was difficult to remove for weighing. In most instances this loss was minor (<5%) and considered acceptable. However, there were several instances within the Fort and Whey Curd natural gravel sites where sediment losses up to 14.6% were observed. These sites contained an abundance of sediment <1 mm and losses >5% could be attributed to an underestimation of the finer grained size fractions during the Malvern granulometer analysis, as observed elsewhere (Agrawal et al., 1991).

#### 2.3.2.3 Quantitative spatial distribution of gravel ≥5 mm within the study site

A continuous gravel survey of the stream-bed was conducted throughout the extent of the study site (Figure 2.1). This was done for two reasons; firstly to determine the naturally available grain-sizes for spawning, and secondly to investigate how rehabilitation had changed spawning gravel availability. Transects across the stream width were taken at approximately 7 m intervals. Three surface gravel samples were taken along the transect at 0.25%, 0.5% and 0.75% channel width using a garden trowel. These were examined on a standard invertebrate sorting tray. Samples were graded in the field using a customised gravelometer; a 200 $mm^2$ polyvinyl chloride (PVC) board with 8 square openings cut out to pass gravel through at 5, 10, 15, 20, 30, 40, 50, 60 mm (Figure 2.4). Notches cut out of the gravelometer edge at 70, 80, 90 and 100 mm were used to grade larger pebbles sizes. The survey data were arranged into 10 mm bin size classes. In order to determine whether rehabilitation gravel had increased surface abundance of suitable salmonid spawning substrate, gravel $64>D\geq16$ mm and $30>D_{50}\geq16$ mm were analysed for increases in abundance between the naturally occurring substrate and a combined natural and *rehabilitated gravel* treatment using the Wilcoxon Signed Rank test for differences. Gravel $60\geq D\geq15$ mm and $30>D_{50}\geq15$ mm were used as proxies for migratory and non-migratory *S. trutta* respectively. This was due to the coarse nature (5 cm) of discrete gravel grading at the sizes surveyed. Non-migratory *S. trutta* gravel size-range was based on the mean length of sexually mature fish within the river (see section 2.3.2.5 below).

**Figure 2.4** The gravelometer used to grade gravel for the continuous streambed survey. Square holes were cut to replicate the sieve mesh shape. Notches along the edges marked 70, 80, 90 and 100 mm.

### 2.3.2.4 Stream velocity associated with sediment distribution

The Water Hall and Whey Curd sites were not included during the 2010 monitoring as access permissions had not yet been obtained. Calculation of the mean velocity at 60% depth and 5 cm above the stream-bed provided a satisfactory indication of near-bed conditions that influenced sediment transport and deposition at spawning sites. Velocities were measured using a Valeport Braystoke BFM002 miniature current flow meter. Revolutions per second (rev $s^{-1}$) were converted to metres per second (m $s^{-1}$) using one of two equations:

Equation 2.2:

$$V = 0.1001n + 0.032$$

where $n$ = rev $s^{-1}$ were in the range of 0.1 - 1.5, and

Equation 2.3:

$V = 0.1079n + 0.030$

where $n$ = rev $s^{-1}$ were in the range of 1.5 - 29.

A designation of velocity useful for site characterisation and comparison was calculated by use of the Froude number (Fr), defined as:

Equation 2.4:

$Fr = V/\sqrt{gD}$

where $V$ = mean velocity at each site, $g$ = acceleration due to gravity and $D$ = mean depth at each site. Velocity was described as tranquil or sub-critical where Fr<1 and turbulent or super-critical where Fr>1 (Hilton and Tordesillas, 2013).

Frequency composition of velocity at each site was examined using $Chi^2$ analysis. Velocities were arranged into the following m $s^{-1}$ bins based on a natural spread of data: 0.01, 0.02, 0.4, 0.08, 0.16, 0.32, 0.64 and 1.28 m $s^{-1}$. Differences between treatments were examined both annually, as well as exclusive of year, and differences within individual treatments were examined between years. Further $Chi^2$ analyses tested whether the introduction of rehabilitation gravel led to an increase in water velocity relative to natural sites. All $Chi^2$ analyses were conducted in Microsoft Excel for Windows 7.

#### 2.3.2.5 Determination of a minimum spawning gravel $D_{50}$ for non-migratory *S. trutta* populations

In order to approximate a $D_{50}$ spawning gravel diameter, a weighted average fish length of all sexually mature non-migratory *S. trutta* was determined based on 10% of the resultant weighted average sexually mature fish length (see Kondolf and Wolman, 1993). The mean age at which fish became sexually mature was identified from a collation of electric fishing survey data collected by the Environment Agency (EA) and the Hull International Fisheries Institute (HIFI), University of Hull. The EA surveys were undertaken in the middle and lower reaches of the river in 2000, 2007, 2008, 2009 and 2011. Surveys in 2011 include fish age data derived from analysed scale samples, whilst all survey years include individual fish length. HIFI fisheries surveys took place during September 2010 and 2011 (Figure 2.5) and included length of individual fish only. A quantitative removal survey method based on Carle and Strub (1978) was adopted by both the EA and HIFI. HIFI sites varied in length from 23 to 87 m covering areas ranging between 83 to 454 $m^2$ (Angelopoulos et al., 2012). Stop nets demarcated the upstream and downstream extent of the sampled river channel and prevented fish moving into or out of the survey site. An Electracatch backpack unit was powered by a 2kVA bank side generator producing 220 V DC output. Three runs through each site enabled construction of depletion curves and fish abundance was estimated from these. Sexual maturity is marked by a sharp decline in the annual growth rate of fish cohorts as greater investment is placed in gonad development than it is in somatic growth (Ricker, 1975). Growth rates for each age class are defined as:

Equation 2.5:

$G_r = \log_e(L_n) - \log_e(L_{n-1})$

where $G_r$ is the growth rate and L is the average length (mm) of fish at age *n* (Ricker, 1975). Weighted average lengths of sexually mature non-migratory *S. trutta* in the River Stiffkey were derived from survey data combined over all years and assigned to 5 cm length class bins. The product of the mid-point of each 5 cm length bin and the frequency of fish within that bin were added and then divided by the sum of the combined frequency data to provide a weighted average length.

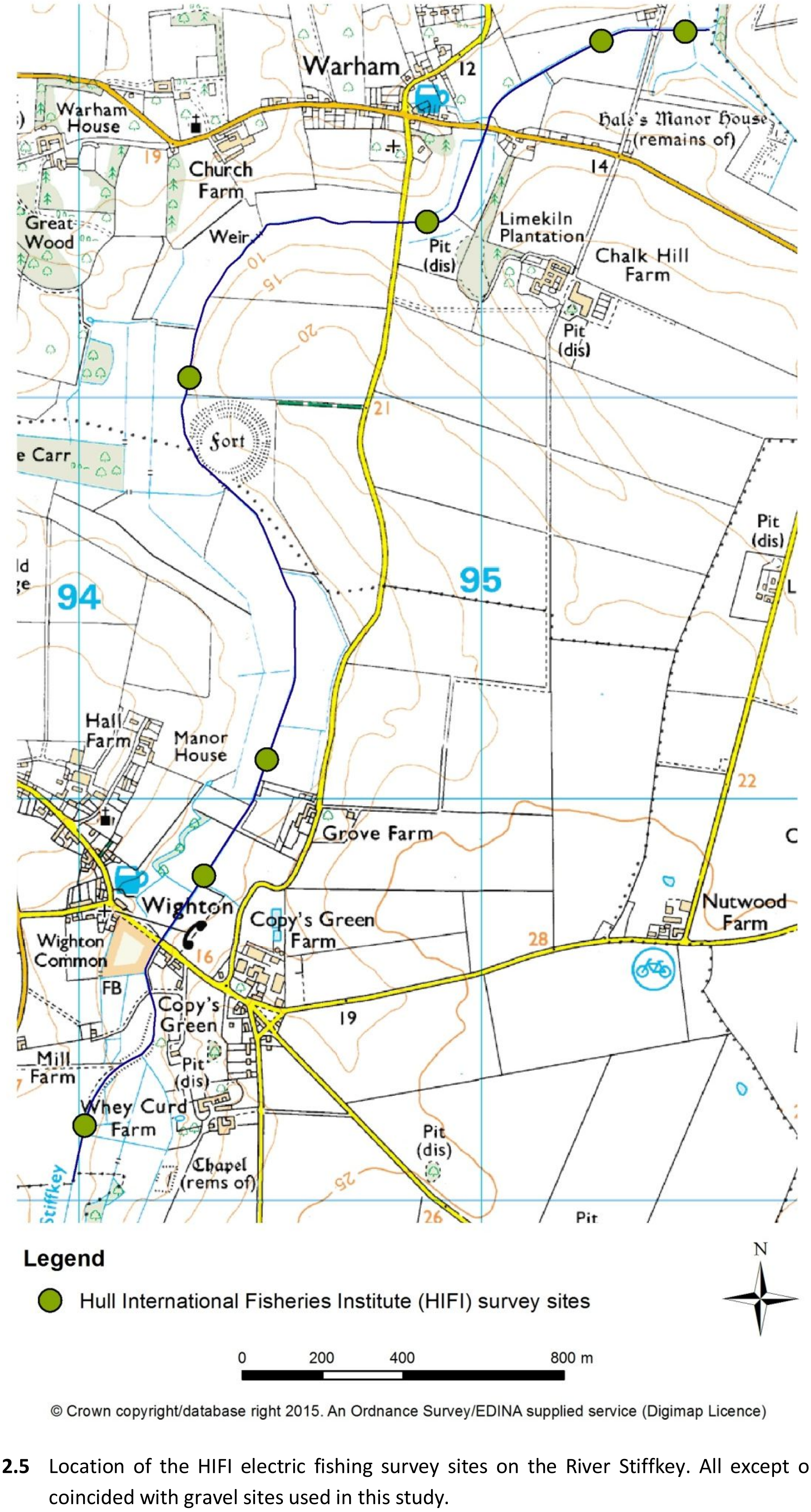



**Figure 2.5** Location of the HIFI electric fishing survey sites on the River Stiffkey. All except one site coincided with gravel sites used in this study.

### 2.3.2.6 Multivariate analysis

Indirect ordination techniques, used to describe sediment composition as well as the effect of velocity on sediment composition between sites, were performed using the software package *Canoco for Windows 4.5* (Lepš and Šmilauer, 2003). Cumulative core grain-size distributions for each site, consistent with Cefas (1999), were analysed in GRADISTAT *v4 for the analysis of unconsolidated sediments* (Blott and Pye, 2001) to determine percentage textural classification descriptors. These were combined into gravel (64>$D$≥2 mm), sand (2>$D$≥0.063 mm) and silt ($D$<0.0063) sediment size-class descriptors, thus ensuring that the number of constrained axes did not exceed the number of sediment size-class variables. Variables were arcsine transformed to avoid problems (see Lepš and Šmilauer, 2003) associated with cumulative percentage values adding up to 100%.

Mean velocity, measured at 5 cm above the stream-bed of each site over three years of monitoring, was used to investigate the relationship with sediment composition. Gradients of sediment data were established along axes which were not constrained by environmental data and therefore provided sedimentological differences between sites. A preliminary detrended correspondence analysis (DCA) was performed in order to test whether the data demonstrated a unimodal or linear response. Second and third ordination axes were detrended by segments to reduce dependence on the first axis. Gradient lengths of the first axis (the largest value) were used to determine whether linear or unimodal techniques would be appropriate to find the largest variability within the data (Lepš and Šmilauer, 2003). Principal components analysis (PCA) is an unconstrained linear ordination method. In order to examine sedimentological relationships (similarity-dissimilarity) between the cores and sites, the scaling was focused on inter-sample distances. Variable scores were divided by their standard deviations to indicate differences in abundance. Data were centred by the sediment classification descriptors. Scaling was focused on the inter-variable correlations to investigate relationships between velocity and sediment. Sediment scores were not divided by their standard deviations (not post-transformed). Variables were standardised to account for the different units of measurement (sediment and velocity).

#### 2.3.2.7 Grain-size distribution statistics

In order to determine granulometric composition variance both within and between sites and treatments, statistical analyses were performed using the Kruskal-Wallis test; a non-parametric equivalent of a one-way ANOVA. Further in-depth analyses were determined using pairwise Mann-Whitney U tests. Cumulative percentage weight of grain-sizes for each 5 cm stratification were analysed at the core-scale. The cumulative percentage weight of cores sampled from each site were used for site-level comparisons. Cumulative percentage sediment weight sampled from sites was used for analysis at the treatment scale consistent with Cefas (1999). In this manner whole core weight differences were standardised and the compositional value of sediments, including cobbles ≥64 mm, were included in the analyses.

Geometric scaling was used within GRADISTAT to determine the cumulative percentile statistics calculated at 5 cm depth intervals for each core: the median grain-size, $D_{50}$, as well as the grain-size at which 10% and 90% of the distribution were smaller, the $D_{10}$ and $D_{90}$. These measurement parameters were based on methods used by Folk and Ward (1957). A physical textural description of each 5 cm core increment was determined based on sediment class (Folk, 1954).

Variance in the depth and frequency of clasts ≥64 mm was determined between treatments. The mid-depth of each clast ≥64 mm within each treatment was analysed using Kruskal-Wallis tests. Post-hoc pairwise Mann-Whitney U analysis determined the difference in depth these occurred in. This analysis indicated the extent of surface gravel redistribution between treatments. The frequency of clasts ≥64 mm within each core were tested with Kruskal-Wallis analysis on a treatment basis.

Average egg deposition depths ranged between 5-20 cm (Crisp and Carling, 1989). The composition of substrate in this zone is essential for successful development of *S. trutta* embryos. This embryo incubation zone was analysed as a distinct feature of the spawning substrate. Cumulative core increments between 5-20 cm of each core for all sites and treatments were analysed for difference in percentage (%) fine sediment (<1 mm). Data were analysed using Kruskal-Wallis tests with pairwise Mann-Whitney U post-hoc analysis. Statistical analyses for specific grain-size ranges: $64>D\geq16$ mm, $30>D_{50}\geq16$ mm, $D<1$mm, $2>D\geq1$ mm, and $D<0.004$ mm were carried out using similar methods outlined above. All statistical analyses were undertaken in Minitab (v16).

### 2.3.3 Estimation of embryo survival from rehabilitation gravels: design overview

An estimate of the egg-to-fry (ETF) survival on rehabilitation gravel was determined in order to assess *S. trutta* spawning quality. ETF estimates were quantified over two consecutive studies. An initial study, conducted in 2011, was aimed at determining the feasibility of egg-box experiments, particularly to verify the required sample size. Five sites were used during the 2011 study: 2003A, 2003C, 2009A, 2009J and Water Hall (Figure 2.1). Water Hall was a naturally occurring gravel site selected for the study based on known salmonid spawning habitat criteria (Crisp and Carling, 1989). Eyed eggs (the stage at which ova within the egg become visible) were installed into the River Stiffkey between 3-4 February 2011. Based on the outcome of the 2011 pilot study, four more sites were included in the subsequent 2012 study to increase replication. In the 2012 study three sites within each treatment provided a balanced study design covering nine sites in total: 2003A, 2003B, 2003C, 2009A, 2009D, 2009J, Whey Curd, Water Hall and Fort. Eyed eggs were installed into the River Stiffkey on 11 and 12 January 2012. Redd surveys were conducted prior to egg installation to ensure minimal disturbance of wild redds.

For each study no depths greater than 50 cm were selected, consistent with known depth limits for *S. trutta* spawning (Kondolf and Wolman, 1993; Armstrong et al., 2003). All sites selected met the minimum current velocity threshold (0.15-0.2 m $s^{-1}$) below which *S. trutta* are thought not to spawn (Louhi et al., 2008; Marchildon et al., 2010). Water velocity was determined at several random locations across each site at 60% depth and at 5 cm above bed substrata to provide an indication of near-bed conditions.

Non-migratory *S. trutta* eggs were used to determine ETF survival. The logistical limitations of obtaining migratory *S. trutta* embryo were great. Furthermore, migratory *S. trutta* depend on the same habitat and water quality parameters during the freshwater phase as non-migratory *S. trutta*. Loss of habitat significantly constrains population recruitment of both species morphs. For these reasons the use of non-migratory *S. trutta* embryo was considered appropriate. *S. trutta* eggs for the 2011 study were obtained from Allenbrook trout farm, Dorset. Females were stripped and eggs were fertilized on 9 December 2010 at the Cefas laboratories, Lowestoft. Eggs were incubated until eyed on 14 January 2011. Due to unseasonably mild temperatures during autumn 2011, eggs from the Allenbrook trout farm were produced earlier than the natural spawning cycle. Therefore eyed *S. trutta* eggs for the 2012 study were obtained from Delfour Hatchery, Scotland, where temperatures were significantly lower thus coinciding with the natural salmonid spawning cycle in Norfolk. Females were stripped and eggs fertilized on 31 October 2011. The eggs eyed on 7 December.


**Figure 2.6** Embryo installation: redd cutting using a HONDA high pressure GX 4-stroke engine pump (a), egg-box installation using the custom made planter to install eyed embryo (b), egg-box ready for installation into a redd (c) and alevin with yolk sac still attached (d).

For both the 2011 and 2012 studies seven redds were cut on each site using a high pressure hose attached to a HONDA GX 4-stroke engine pump with a maximum output capacity of 500 l $min^{-1}$ (Figure 2.6a). Effort was standardised at 15 mins per redd. Working in an upstream fashion to minimise sediment disturbance, the natural redd cutting method was mimicked ensuring redd dimensions adhered as closely as possible to those outlined in Crisp and Carling (1989). Redds were arranged on each site from 1-7 so that 1 was the upstream-most and 7 the downstream-most redd (Figure 2.7a). This layout ensured good site coverage. Four egg-boxes were installed into each redd (200 eggs per redd, 1400 eggs per site). Egg-boxes were arranged in a square formation, approximately 15-20 cm apart (Figure 2.7b). Egg-boxes were made from fine PVC Netlon mesh and were cylindrical in shape, 10x5 cm (Figure 2.6c), consistent with Harris (1972). Each box was partially filled with gravel collected from each respective site and washed to remove the fine sediment fraction. 50 eyed eggs were introduced into each egg-box and overlaid with more gravel. Lids were securely wired in place. The small aperture size of the netting prevented escape of hatched alevins and fry but was large enough to ensure that eggs were exposed to stream hydrology and sediment conditions, thus ensuring a direct calculation of survival. Egg-boxes do not bias the egg samples in any way through the accumulation of detrimental fine sediments or encouragement of fungal infection (Harris, 1973). A good indication of survival under natural conditions was therefore achieved. Installation of egg-boxes into redds was facilitated by a custom-built egg-box planter; a hollow steel tube into which a steel spike was fitted and used to drive the tube into the gravel. Removal of the steel spike accommodated the passage of egg-boxes through the steel tube and into the gravel deposit (Figure 2.6b). Egg-boxes were buried to a mid-depth of 13 cm so that eggs within the egg-boxes (H: 10 cm) were distributed within a range of 8-18 cm, consistent with burial depths observed by Crisp and Carling (1989), 5-20 cm.

Other work estimating ETF survival had comparable methodologies. A similar egg-box study design with PVC Netlon egg-boxes was employed by Harris (1972), however that study design differed by inserting egg-boxes into substrate adjacent to natural redds. Egg-boxes of comparable design (9x6.5 cm) each filled with 50 embryo and gravel (range 5-30 mm) were used in an ETF study conducted by Syrjänen et al. (2008), 4-6 egg-boxes were placed in baskets that were in turn buried within incubation gravel. Dumas and Marty (2006) inserted 10 Atlantic salmon (*Salmo salar*) eggs into fine mesh screen cylinders, 12 $cm^3$, and installed 400 eggs in each redd, consistent with this study. Rubin (1995) used two sizes of PVC egg-boxes cylinders, 7x16 cm and 7x10 cm, filled with 200 and 100 eggs in each respectively. That study however differed as it did not include PVC lids but a window covered with PVC to seal the unit and provide access (Rubin, 1995).

The incubation period of *S. trutta* embryos are temperature dependent (Harris, 1973; Crisp, 1981; Crisp, 1988). Temperatures of river water from the EA Warham Gauging Station were used to calculate degree days for hatching and first-feeding. A simple model based on known egg incubation periods at prescribed temperatures enabled prediction of the hatching and first-feeding stages (Crisp, 1988). Temperatures were monitored from 9 December 2010 to 25 March 2011, and again from 31 October 2011 to 19 March 2012 using 3 surface water temperature probes distributed throughout the study site. Intragravel water temperatures within redds can however be on average 0.5°C warmer than surface water (Sear et al., 2014). Embryos were recovered on 25 March 2011 and 19 March 2012. Egg-boxes were carefully dug-up using a garden trowel in an upstream fashion to prevent disturbing eggs left in situ. Egg-boxes were immediately transferred to a bucket of clean river water and sorted on white trays. Alevin (dead and alive), fry (dead and alive), recent-dead eggs, long-dead eggs and presence of leeches were recorded. Degree of fungal growth and sediment intrusion were noted. Live alevin and fry were dispatched under Schedule 1 of the Home Office Animal (Scientific Procedures) Act 1986. Survival rates were calculated directly from the ratio of recovered alevins and fry, dead or alive, to the total numbers of eggs planted.

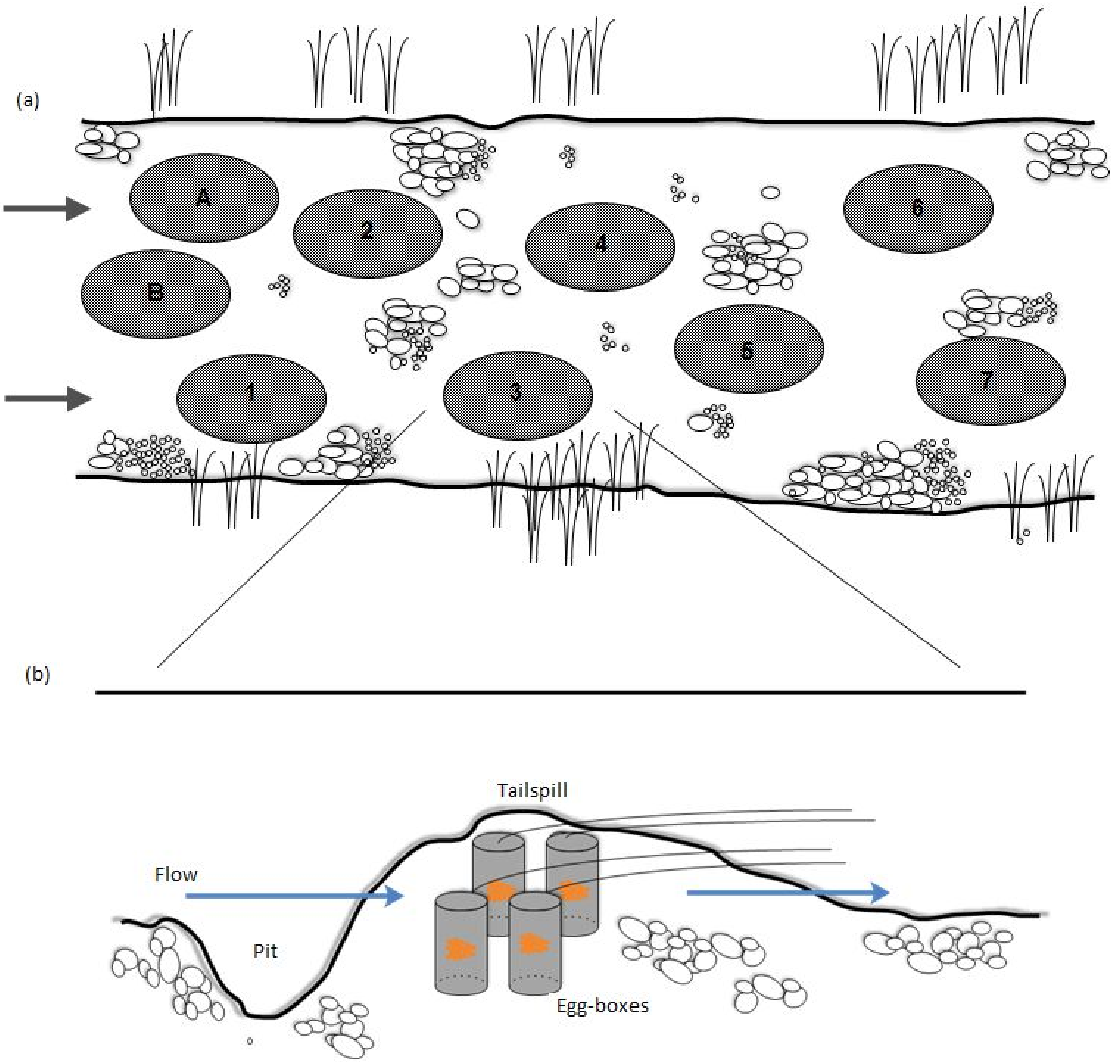



**Figure 2.7** Schematic of the location of individual redds per site, 1-7 (a), and profile view of egg-boxes installed within redds (b), illustrating the egg-box experiment design on each site and within each redd. A and B indicate redds cut for gravel coring purposes, post- and pre-incubation respectively. Note the cord attached to ensure retrieval of egg-boxes.

Water velocities at several points around each redd were recorded consistent with the procedures adopted by Crisp and Carling (1989). Points directly in front of and to the sides of the redd pit, as well as on top of the highest point on the tailspill at 0.6 x depth (Figure 2.8). Velocity was measured just after installation and prior to egg retrieval using a Valeport Braystoke BFM002 miniature current flow meter. Finances prevented the use of expensive interstitial velocity sensors.

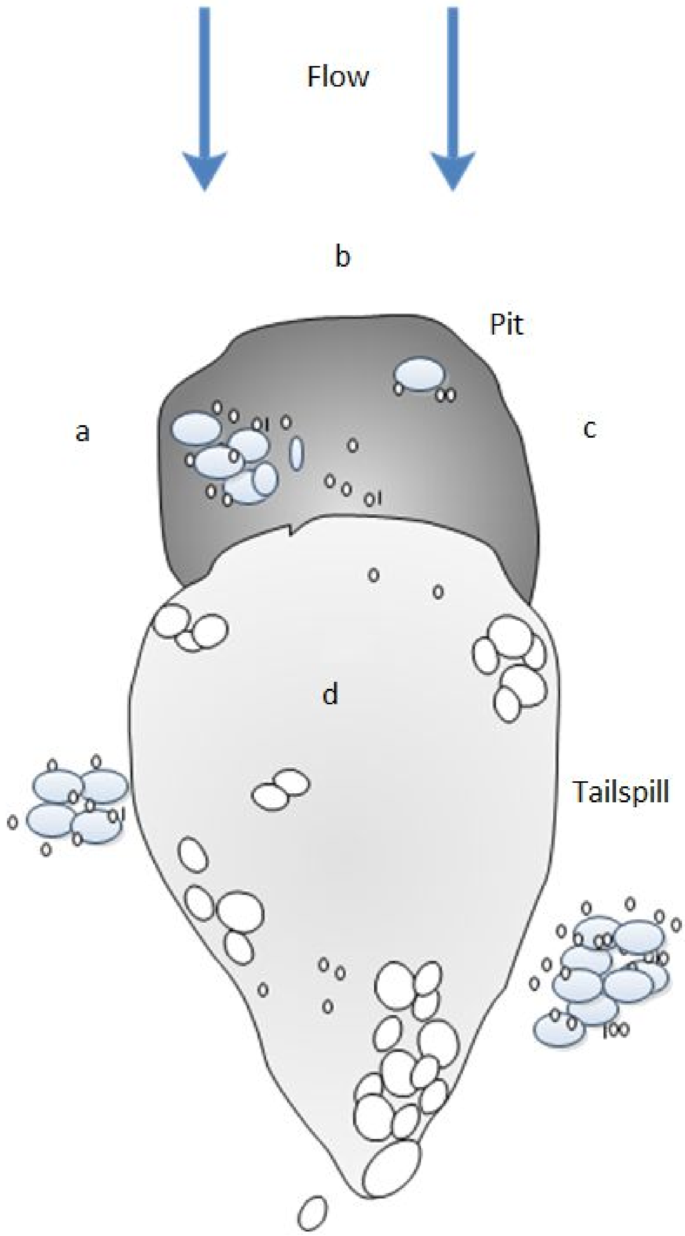



**Figure 2.8** Location of redd flow measurements points: a, b and c are located at points around the pit, and d located at the highest point on the tailspill. Arrows indicate the direction of water flow.

Additional redds were cut for gravel freeze coring, pre- and post-incubation, during the 2012 study only (Figure 2.7a). Pre-incubation redds were cut on the natural gravels at Whey Curd, and restoration sites 2003A and 2009A, the upstream-most site of each treatment. These redd substrata were sampled to provide baseline sediment conditions into which embryo were installed. Low fine sediment (<1 mm) variability between individual redds within each treatment was assumed based on loss of the fine sediment fraction during the redd cutting process. Additionally the pre-incubation freeze core redds illustrated how effectively sediment

<1 mm was removed from bed substrata during the redd cutting process. Post-incubation redds were cut on all sites used for the study. These were cored immediately after egg recovery and provided an indication of sediment accumulation during the embryo incubation period. The Whey Curd, 2003A and 2009A sites had a total of nine redds whilst all other sites had a total of eight redds. Dumas and Marty (2006) followed a similar redd grain-size composition sample design, however they used a rectangular metal bucket (30x20x20 cm) as opposed to freeze coring. Although Pulg et al. (2013) sampled redd grain-size characteristics by means of freeze cores, they sampled at the end of embryo incubation only.

#### 2.3.3.1 Statistical analysis of embryo survival

The egg-to-fry survival histogram was initially analysed for normality using an Anderson-Darling test. Then, Kruskal-Wallis tests, which have a lower probability of making a type I error (less likely to find a significant result when there is not one) (Fowler et al., 2009), were used to analyse egg-to-fry survival rates of treatments and sites for distributions displaying similar medians. Post-hoc analysis was performed using non-parametric pairwise Mann-Whitney U tests.

Grain-size distribution statistics were derived from GRADISTAT, as described above in section 2.3.2.7. Grain-size composition within the redd environment is fundamental to embryo survival. Tests for difference between treatment redd sediment composition before embryos were installed provided an indication of the relative sediment incubation environment. Kruskal-Wallis tests were carried out on the cumulative percentage weight data, including clasts ≥64 mm, to determine differences between pre-incubation redd gravel cores. Post-hoc analysis were conducted using pairwise Mann-Whitney *U* tests. Difference in composition between specific grain-size ranges present in redds: $64>D\geq16$ mm, $30>D_{50}\geq16$ mm, $D<1$mm, $2>D\geq1$ mm, and $D<0.004$ mm was determined using Kruskal-Wallis analysis.

The sediment composition within the embryo incubation zone, 5-20 cm, has a direct effect on embryo development. The median cumulative percentage weight of fine sediment <1 mm ingressed during embryo incubation was assessed for the incubation period. A threshold limit of <1 mm for fine sediment has been successfully employed in other salmonid ETF survival studies (see Dumas and Marty, 2006; Heywood and Walling, 2007). Dumas and Marty (2006) observed an increase in sediment <1 mm in redds post embryo incubation. This increase was associated with poor ETF survival through decreased interstitial permeability and reduced DO concentration and saturation levels. Fine sediment <1 mm has also been reported a suitable quality index for salmonid spawning gravel habitat (Scott and Beaumont, 1994; Kondolf, 2000), and includes silt and clay size fractions each shown to have detrimental effects on developing embryo (Greig et al., 2005b). McNeil and Ahnell (1964), Cederholm and Salo (1979) and Tagart (1984) indicated that sediment <1 mm was responsible for reduction in spawning substrate permeability. Composition of coarse sand ($2>D\geq1$ mm) within surface 10 cm of substrata was also examined. The SI, used to measure the composition of sand in spawning substrate as a proxy for the ability of fry to emerge from spawning gravels, was calculated for 5-20 cm substrate depth.

Percentage frequency velocity of the embryo-redds and freeze cored-redds were analysed for treatment differences between years, within years, and irrespective of study years by means of $Chi^2$ tests. Bins used where: 0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, 1 and >1. Froude numbers were calculated to characterise stream flow over spawning gravel sites. Statistical analyses were undertaken in Minitab (v16), except for $Chi^2$ analysis (as used above) which was conducted in Microsoft Excel for windows 7.

Indirect ordination techniques, used to describe the relationship between sediment composition and velocity, were performed using *Canoco for Windows v4.5*. The 2011 feasibility study had no associated sediment data and as such no ordination tests were conducted for this study. Mean velocity from the 2012 study were used exclusively. Cumulative grain-size distributions within the embryo incubation zone, 5-20 cm core depth, were analysed. No post-incubation core could be sampled for the 2009J rehabilitation gravel site. As such post-incubation redd sediment composition for this site was derived from median values of 2009A and 2009D redd substrate. The preparation of sediment data followed the method for multivariate analysis outlined above (section 2.3.2.7) for the 2012 study only. A preliminary DCA was performed in order to test whether the data demonstrated a unimodal or linear response. Second and third ordination axes were detrended by segments to reduce dependence on the first axis. Gradient lengths of the first axis (the largest value) were used to determine whether linear or unimodal techniques would be appropriate to finding the largest variability within the data (Lepš and Šmilauer, 2003). In order to examine the relationship between sediment and velocity, the scaling was focused on inter-variable correlations. Variables were standardised to account for the different units of measurement within the data set.

### 2.3.4 Quantitative spatial analysis of juvenile *S. trutta* life-stage dependent habitat

A qualitative assessment of juvenile *S. trutta* habitat based on the APEM continuous mapping walkover survey method was conducted 6-9 July 2011. As a continuous survey, this method differed from the River Habitat Survey (RHS), which is focused on discrete river lengths (500 m) (Fox et al., 1998). Further, the RHS is designed around the presence/absence of 200 compulsory features. The survey conducted in this study identified the presence of suitable *S. trutta* habitat, preferentially identifying juvenile habitat and flow biotopes (in-stream parameters) only. This survey provided a snapshot of habitat availability at a given point in time. The river outline and associated prominent geographical features were selected from high resolution 1 $km^2$ digital tiles of the Ordnance Survey MasterMap 1:1 000 series and printed on A3 waterproof paper. Each map was geographically referenced and overlaid with the Ordnance Survey National Grid. Approximately 300-350 meters of river illustrated in each map provided sufficient detail to accurately survey relevant and predefined habitat characteristics. Observed in-stream habitat features and river flow characteristics were drawn and annotated directly onto the maps (Figure 2.9). Actual habitat position and percentage habitat cover were approximated as accurately as possible through a combination of readings from a hand held GPS, mapped Ordnance Survey British National Grid (BNG) lines and salient geographical features. Maps annotated in this manner provided a high level of detail not attainable from alternative survey methods.

Physical habitat characteristics associated with key stages of juvenile *S. trutta* life history: nursery habitat (area of early development used for first summer), rearing habitat (autumn and winter habitat) and overwintering habitat were identified (Table 2.2). Specific habitat types identified included: undercut banks, overwintering refugia, high flow fry refugia, large woody debris, vegetation stands, marginal habitat, overhanging vegetation, and rehabilitation gravels. Discrete identification of all natural spawning gravel habitat was not conducted due to high spatial and temporal sediment composition variability of the streambed. Given such variability, gravel streambed habitat, a matrix-filled coarse gravel substrate that included spatially variable natural spawning habitat, was surveyed. These habitat types cover refuge requirements during the first year of *S. trutta* life. Due to the modified character of the channel, stream flow was mostly homogenous across the stream width. Alternative flow biotopes were therefore defined by visual observation of surface flow types consistent with Padmore (1998) and stream depth: run (<30 cm stream depth), glide (30-60 cm), deep glide (60-90 cm), very deep glide (90-120 cm) or pool (>120 cm). Stream depth determined the characteristic nature of each flow biotope: runs were shallow faster water, whilst a glide had smooth consistent slow flow with little surface undulations. Individual A3 survey maps were scanned and digitised in ArcMap (v10.2) (Figure 2.10). Each map was georeferenced to Ordnance Survey MasterMap 1:1 000. A 1st order polynomial transformation was used in each instance as no warping of survey maps was required. A bilinear interpolation was used although either nearest neighbour or cubic interpolation would have been sufficient as survey maps were derived from the base map.


**Figure 2.9** An example of the Ordnance Survey MasterMap 1:1 000 series map illustrating river channel and annotated habitat features. Maps were printed on A3 waterproof paper for use in the field. These provide a high level of detail not obtained through other methods. Inset indicates annotation details.

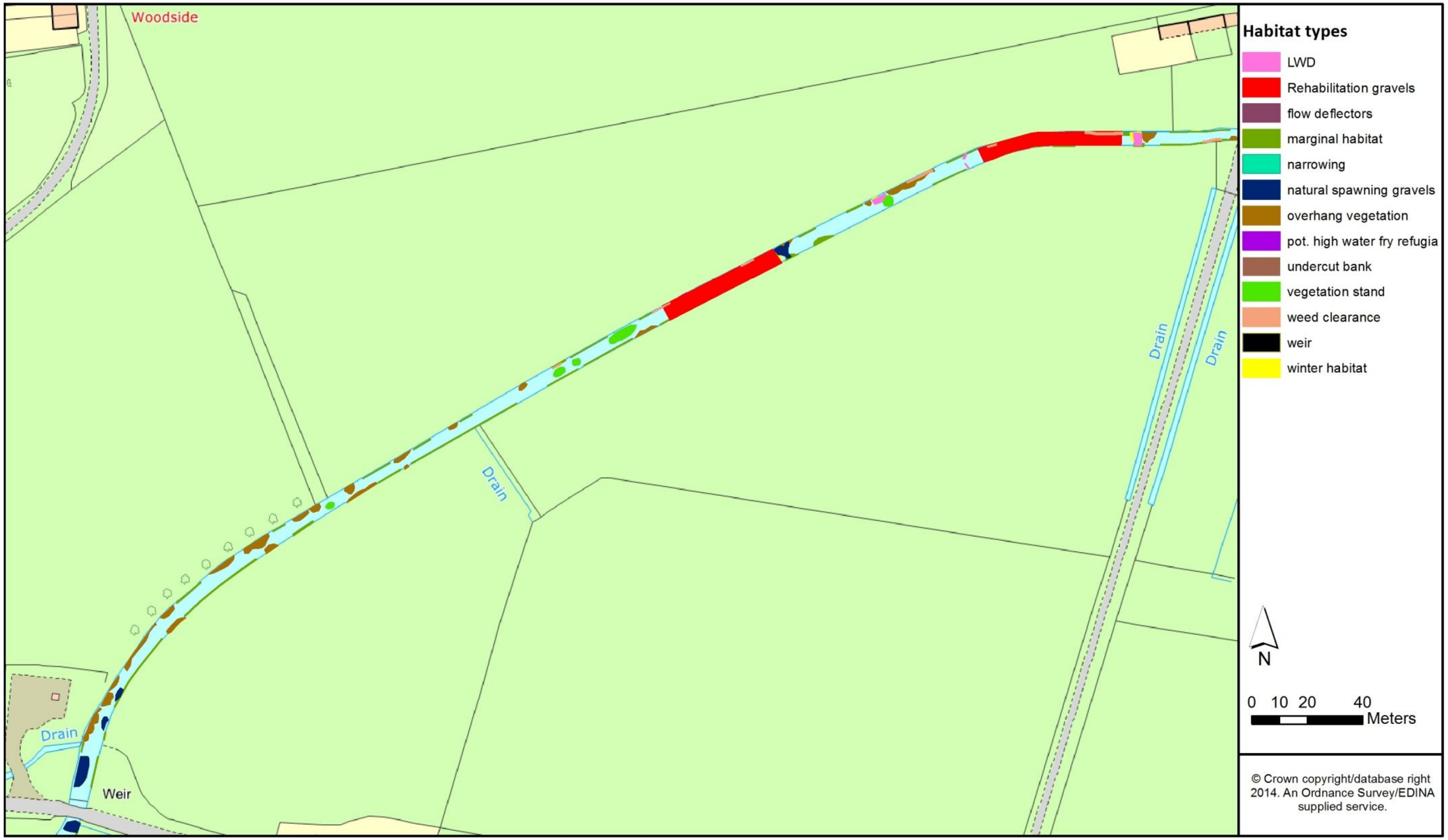



**Figure 2.10** An example of the digitised habitat types in the river channel based on Figure 2.9. Digitising habitat types in this manner enabled quantitative and spatial analysis of early life-stage dependent data.

**Table 2.2** Summary of habitat types for juvenile life-stages of *S. trutta*. Spatial fragmentation between these habitat types impedes population recruitment.

| Habitat type | Habitat descriptor | Reference |
|---|---|---|
| Spawning | Velocity: 0.2 - 0.80 $ms^{-1}$ | Crisp & Carling (1989), Barnard & Wyatt (1995), Louhi et al. (2008) |
| | Depth: 60-820 mm | Crisp & Carling (1989) |
| | Gravel size: 64>*D*≥16 mm | Crisp & Carling (1989), Louhi et al. (2008) |
| | Low fines content | Crisp & Carling (1989), Hendry et al. (2003) |
| Nursery | Velocity: 0-0.2 $ms^{-1}$ | Bachman (1984), Crisp & Hurley (1991), Bardonnet and Heland (1994), Greenberg (1994), Heggenes et al. (1999) |
| | Depth: 50-300 mm | Jonsson (1989), Bardonnet and Heland (1994), Greenberg (1994), Heggenes et al. (1999) |
| | Substrate: 10-90 mm | Heggenes (1988b), Bardonnet and Heland (1994),Heggenes et al., (1999), Hendry et al.(2003) |
| | Channel margin habitat | Greenberg (1994) |
| Parr | Velocity: ≤0.70 $ms^{-1}$ | Bachman (1984), Heggenes (1988a), Crisp (1993), Heggenes (1996) |
| | Depth: 140-1220 mm | Heggenes (1988a), Maki-Petays et al (1997), Hendry et al. (2003) |
| | Substrate: 60-250 mm | Heggenes (1988a), Heggenes et al. (1999) |
| Over-wintering | Substrate: 60-500 mm | Heggenes (1988b), Heggenes et al. (1993), Alfredsen & Tesaker (2002), Annear et al. (2002) |
| | Low degree embededness | Heggenes et al. (1993), Alfredsen & Tesaker (2002), Armstrong et al. (2003) |
| | Access to deeper, slower water | Heggenes et al. (1993), Alfredsen & Tesaker (2002), Annear et al. (2002) |
| | Dense brushy marginal areas. | Heggenes et al. (1993), Alfredsen & Tesaker (2002) |

#### 2.3.4.1 Delineation of juvenile production zones within the study site

An investigation into potential *S. trutta* production zones within the River Stiffkey were examined based on juvenile life-stage dependent habitat. Functional habitat units (FHU), areas ($m^2$) that contained habitat suitable for *S. trutta* production, were identified consistent with Kocik and Ferreri (1997). Kocik and Ferreri (1997) propose that a river be divided into FHU determined by the spatial fragmentation of key life-stage habitat. The limits of each FHU are defined by the dispersal ability or migration capacity expected for the life-stage under investigation. For example, an FHU of recently emerged fry is determined by the extent of fry dispersal after emergence and the discrete habitat area ($m^2$) within that range. There is greater *S. trutta* production in stream sections with high abundance of spatially connected life-stage habitat (Kocik and Ferreri, 1997). Subsequently, areas of juvenile *S. trutta* production can be geographically located along the length of the river for each life-stage. FHU's illustrated in this manner offer an invaluable tool for river management and provide a natural scale, defined by the physical attributes of the stream, that inform management decisions.

FHU are constrained by the maximum juvenile dispersal distances between key habitat types. Two critical migrations linked to juvenile survival were examined: firstly, between spawning and nursery habitat, and secondly, dispersal from rearing to overwintering habitat. Migration distances between spawning and nursery habitat were associated with stream velocity (Ottaway and Clarke, 1981; Ottaway and Forrest, 1983; Elliott, 1987). Based on studies conducted by Elliott (1981; 1987) and Ottaway and Clarke (1982), dispersal distances travelled by fry emerging from spawning gravels to nursery habitat ranged from 10 m to 40 m under associated stream velocities of 0.1 m $s^{-1}$ to 0.5 m $s^{-1}$ respectively. Juvenile dispersal from rearing habitat to establish overwintering refuge during autumn was examined at a distance of 100 m consistent with Solomon and Templeton (1976) and Brown et al. (2001). These authors concluded that >50% of $0^+$ *S. trutta* in a chalk stream remained within 100 m of the spawning gravels they emerged from. Key life-stage habitat relocations in chalk streams have been observed to be mostly (>90%) in a downstream direction (Solomon and Templeton, 1976; Moore and Scott, 1988; Daufresne et al., 2005).

FHU in this study were derived in the same manner. Spatial analysis was conducted in ArcMap (v10.2). FHU were produced by extracting a buffer of the maximum juvenile dispersal distances at each of the examined life-stages from the digitised habitat survey data. Distances were measured downstream from rehabilitation gravels as well as natural gravel sites. Where buffers touched or overlapped another FHU they were aggregated into a single FHU. The greater the area ($m^2$) the greater the *S. trutta* production potential. If buffers did not include the recipient habitat type then they were not considered. Longitudinal river length was associated with each FHU. This was measured from the upstream most point of the study site. The combined area ($m^2$) of life-stage habitat within each FHU was plotted against the longitudinal river length.

#### 2.3.4.2 Quantitative survey of marginal habitat loss

It became apparent in March 2012 that a large amount of marginal bank vegetation had been removed from the river channel throughout the Holkham Estate reach of the study site (Figure 2.1). Drought conditions across East Anglia during the winter and spring months were responsible for lower than average river levels. It is likely that farmers removed the marginal vegetation in order to increase water flow to a surface abstraction pump further downstream. After emerging from incubation gravel, fry seek refuge in shallow marginal habitat and much of the removed vegetation was indeed fry rearing habitat expected to have been occupied at this time of year.

An impromptu survey quantified the extent of habitat removal throughout the study site. Using a handheld GPS, the area ($m^2$) of habitat removal was annotated onto habitat survey maps (Figure 2.9). Loss of marginal vegetation was considered against the continuous habitat survey conducted 6-9 July 2011 (section 2.3.4 above) and within the analysis of FHU. Where marginal vegetation had been removed in the landscape, a representative area was removed from the habitat survey to reflect more accurately a snapshot in time of the available habitat for juvenile *S. trutta*.

# 3 Catchment controls: historical perspectives and contemporary dilemma

## 3.1 Introduction

Climate, local lithology, channel gradient, land-use, stream sinuosity and catchment topography control hydrogeomorphic processes (Knighton, 1984; Wharton, 2000). These processes, in turn, determine river bed morphology, reach characteristics, bedform variability and ultimately the availability and suitability of salmonid spawning habitat over the medium- to long-term (Knighton, 1984; Kondolf and Wolman, 1993; Milan et al., 2000; Hendry et al., 2003; Greig et al. 2005a; Moir and Pasternack, 2010). Anthropogenic influences within the catchment significantly alter these natural interactions, changing the physical structure of spawning habitats over the short-term. The River Stiffkey, like most European rivers, has been modified, predominantly through flood regulation management and land-use practices, with concomitant impacts on salmonid ecology.

This chapter investigates catchment processes and upstream controls of the River Stiffkey, particularly in terms of catchment geomorphology and its role in defining morphosedimentary conditions and the physical context for rehabilitation gravel characteristics. Key objectives are:

- determine key catchment controls and hydrogeomorphic processes that regulate the physical character of the river, including the rehabilitation gravels
- establish the cause/s of excessive sedimentation.

These factors underpin key physical constraints to *S. trutta* recruitment in the River Stiffkey at a catchment scale. Introduction of rehabilitation gravels to the river channel have increased habitat heterogeneity at a macrohabitat scale but largely fail to address dominant morphological control and processes that operate at a larger scale. It is these larger scale processes that ultimately define the physical suitability of rehabilitation gravels for *S. trutta* recruitment. This chapter reports on a desktop based study using primarily GIS analysis (ArcMap v10.2). Historic and contemporary land-use, precipitation and discharge data as well as the dominant geology and the longitudinal profile of the river channel at the catchment scale are investigated.

## 3.2 Catchment controls on river geomorphology

The River Stiffkey shares many characteristics with those associated with a typical chalk stream; it has a light-bulb shaped catchment due to groundwater-sapping (near constant drainage from a single fixed point) typical of groundwater-fed drainage basins (Schumm et al. 1995), with a low drainage density. A third order stream, the River Stiffkey drains predominantly agricultural land with a long history of modification. The hydraulic regime is an important internal control on channel behaviour and modification. It has a seasonally variable character determined by climate (frequency and magnitude of precipitation), geological characteristics, catchment morphology, vegetation, channel characteristics and catchment land-use (Leopold et al., 1964; Charlton, 2008). Climate, geology and catchment topography provide the energy that drives water and sediment within the catchment (Leopold et al., 1964; Easterbrook, 1999). Based on the internal control mechanisms within the River Stiffkey catchment, channel stream power, the potential energy to impart physical channel form change determined by the channel gradient and discharge, is poor.

### 3.2.1 Catchment topography and geology

The area drained by the River Stiffkey, approximately 140 km$^2$, is the largest catchment in North Norfolk. Catchment topography is dominated by low relief (Figure 3.1) that creates a gradual gradient throughout the length of river. An incision valley is apparent in the mid-reaches of the River Stiffkey that represents the greatest slope angles in the catchment (Figure 3.1, see inset map), but this has little effect on stream gradient (longitudinal section) measuring a slope of 1:400 through this area. Hillslopes in the western catchment are greater than the eastern. The dominance of gentle sloping valley hills in the eastern part of the catchment has moderate run-off delivery relative to the western divide.

Catchment bedrock geology is dominated by Cretaceous Chalk (Figure 3.2). Long water residency times in chalk aquifers ensure stream velocity has a high degree of constancy over time. Chalk aquifers, therefore, typically moderate stream discharge and maintain base flows throughout drier months (Berrie, 1992; Sear et al., 1999). In the same manner, however, flood hydrographs are also moderated, and peak flow magnitudes in response to precipitation within the catchment are reduced (Sear et al., 1999). The long water residency times in chalk aquifers also maintain a stable thermal regime, between 5-17° C (Mackey and Berrie, 1991).

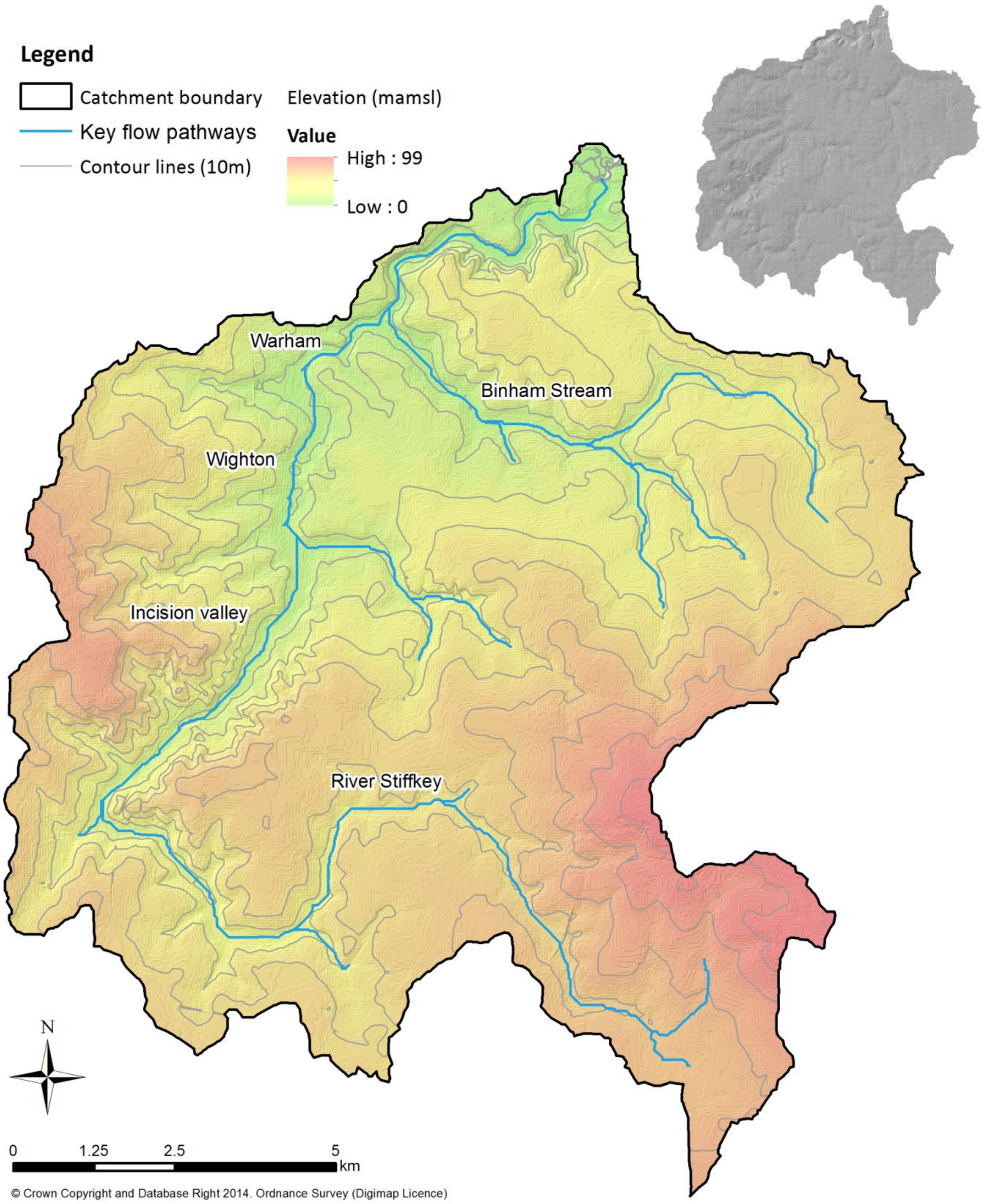



**Figure 3.1** Topography of the River Stiffkey catchment, characterised by low lying relief. An incision valley within the mid-reaches of the river, observed within the western divide, is the steepest landform relief.

Comparatively, upland rivers have a much wider temperature range, -1-23° C, due largely to external atmospheric factors such as air temperature, snow melt, precipitation and discharge characteristics (Smith and Lavis, 1975; Webb and Walling, 1992). Although the chalk geology is the principal unit of stream hydrology, which provides the majority (76%) of baseflow to the River Stiffkey (Environment Agency, 2013), the rest flows off glacial deposits in the east of the catchment. The chalk bedrock is overlaid in the eastern divide by clay, silt, sand and gravel deposits of the Sheringham Cliffs Formation. These sediments are rich in chalk and flint (Marly Drift) due to the action of glacial retreat over a chalk dominated bedrock (Hiscock et al., 1996; Holman et al., 1999; Ander et al., 2006). This superficial geology facilitates a large supply of poorly sorted sediment within the river catchment. The upper river reaches flow through very poorly sorted diamicton of Mid-Pleistocene glacial origin that comprise a wide clast size range. Diamicton deposits alter the hydrodynamic nature of the River Stiffkey in the upper reaches. Here, the Palaeogene Clays prevent recharge to the underlying chalk bedrock causing precipitation to permeate through the glacial deposits and increase the characteristic chalk bedrock dominated response time to precipitation (see section 1.8, Chapter 1).

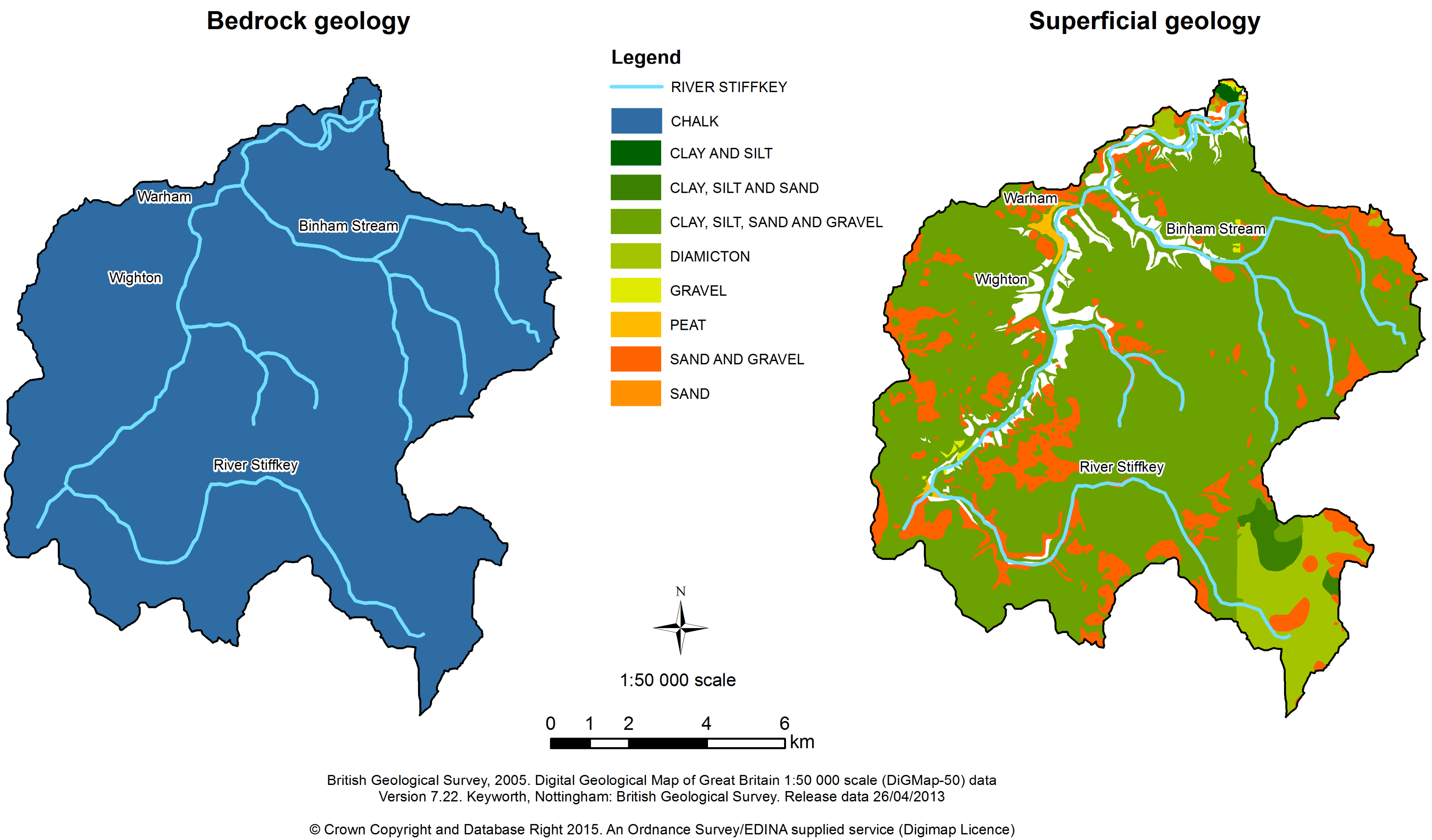



**Figure 3.2** Bedrock and superficial geology of the River Stiffkey catchment. The hydrodynamic nature of the River Stiffkey was determined by both the moderating effect of the chalk aquifer and the overlying superficial deposits which serve to increase the response period to precipitation.

### 3.2.2 The influence of precipitation on stream discharge and sedimentation

East Anglia is the driest region in the UK (Environment Agency, 2005). The River Stiffkey catchment received 55.6 mm monthly mean rainfall and a mean annual of 666.8 mm between 1910-2011 based on 5x5 km grid catchment precipitation data from UKCP09 (Jenkins et al., 2009). Intra-annual variability of mean rainfall during this period was low, however the range of values particularly over summer months was high (Figure 3.3). Annual trend of mean monthly rainfall increased from spring (45.4 mm) through to autumn (63.3 mm), and then decreased over winter (mean 55 mm) (Figure 3.3a). Rainfall during the winter months was more consistent, observed as a lower range of data spread around the median during these months. Although summer rainfall was greater (mean 58.7 mm), it was less regular with greater variation around the median.

In some years the 6 consecutive months of summer through to autumn could represent the wettest period of the year as intense convective precipitation is common, however the minima in the summary boxplots shows that this was not always the case and indeed the inverse was true for some years. The long-term trend of rainfall has remained similar over time with no significant change between the period 1910-1971 and the period during which discharge data from the Warham EA gauging station was measured, 1972-2011 (Figure 3.3b and c). Mean monthly rainfall variability was relatively high over the annual scale, frequently varying by 100-200 mm between years. Trend analysis over 10 and 25 years, however, indicated little change over the long term (Figure 3.4 and 3.5a). There is therefore no evidence of a climate-driven change in flow regime between 1972-2011 (Figure 3.5b and c) and as such any alteration was likely a reflection of the historically modified character of the stream channel (see section 3.3.2) not observed in the current time-series of discharge data.

Daily discharge between 1972-2013, measured at the Environment Agency gauging station near the village of Warham located towards the downstream end of study site (Figure 2.1), reflected an annual oscillation cycle of high winter and low summer flows (Figure 3.6), typical of a chalk stream. Regular annual peak discharges reflected the glacial response to high rainfall in the upper reaches. However, the 5 year trend indicated a more consistent discharge over the long-term (Figure 3.6). Long-term discharge variability was characteristically low with a mean of 0.58 $m^3 s^{-1}$, interspersed with regular annual high discharge events (S.D. = 0.56 $m^3 s^{-1}$) (Figure 3.7). Frequency of daily discharge ≤0.8 $m^3 s^{-1}$ was high, whilst high discharge events >3.2$m^3 s^{-1}$ were rare. Daily discharge >1 $m^3 s^{-1}$ was not common (figure 3.8). Flow during 1975 and 2007 had over 50 days of discharge >1 $m^3$ $s^{-1}$. However, most years had <20 days of discharge >1$m^3$ $s^{-1}$.

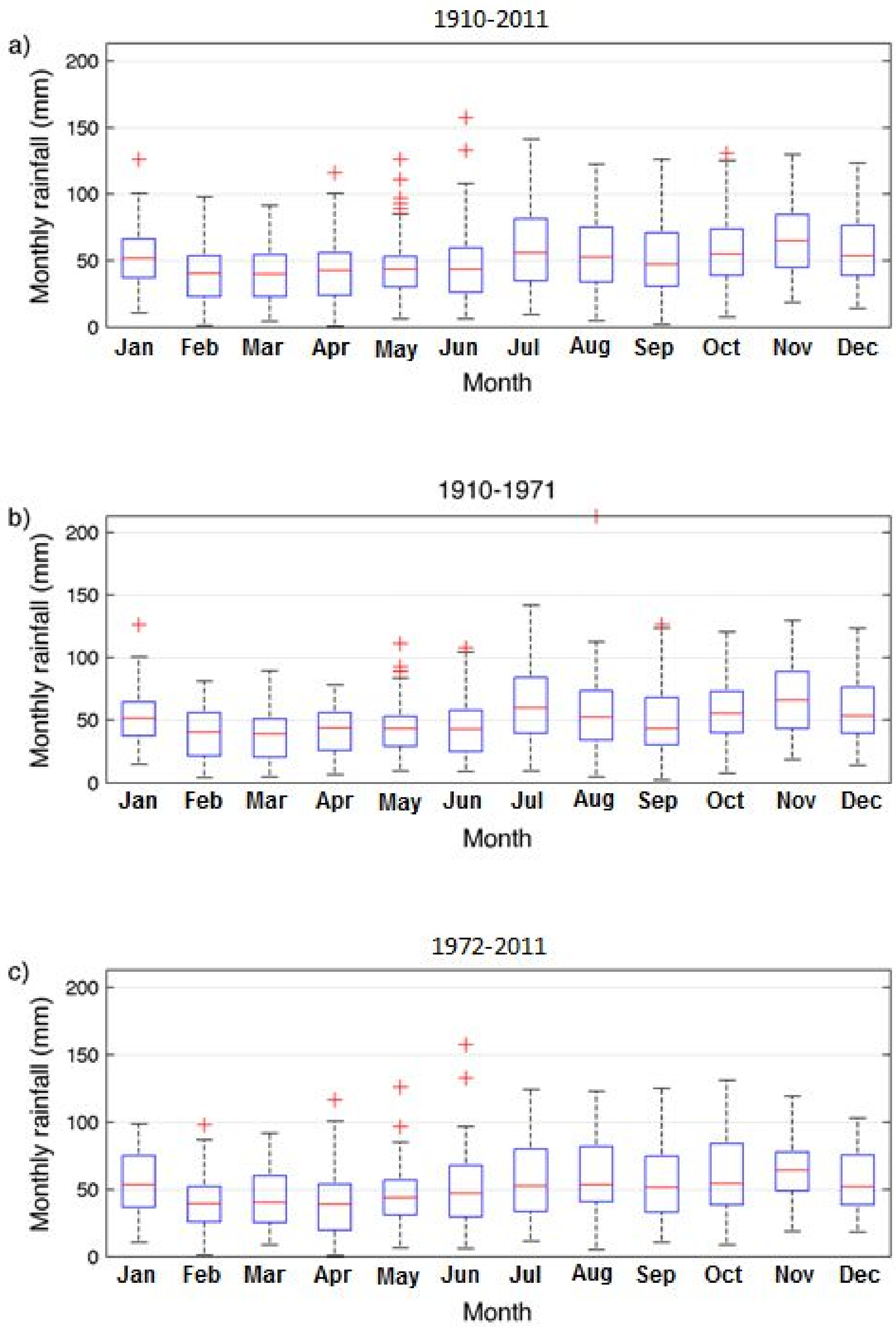



**Figure 3.3** Summary boxplots of monthly rainfall (mm) time-series from 1910-2011 (a), 1910-1971 (b), and 1971-2011 (c). Winter rainfall was more consistent relative to the sporadic high magnitude nature of summer and autumn months. There was little difference between 1910-1971 and 1972-2011 rainfall. Time-series data were collated from a 5x5 km grid used for the UKCP09 (Jenkins et al., 2009).

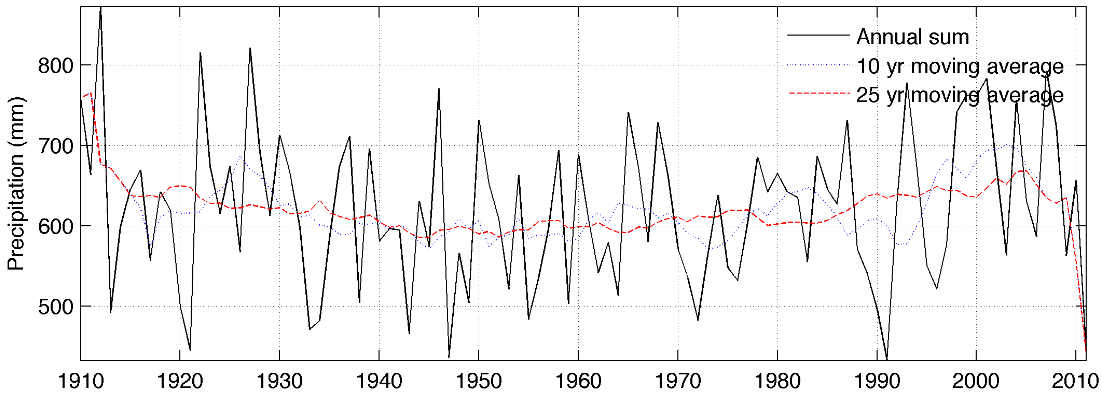



**Figure 3.4** Annual sum precipitation time-series plot 1910-2011 with 10 year and 25 year moving averages. Annual variability was high, however, variability over the time-series remained low.

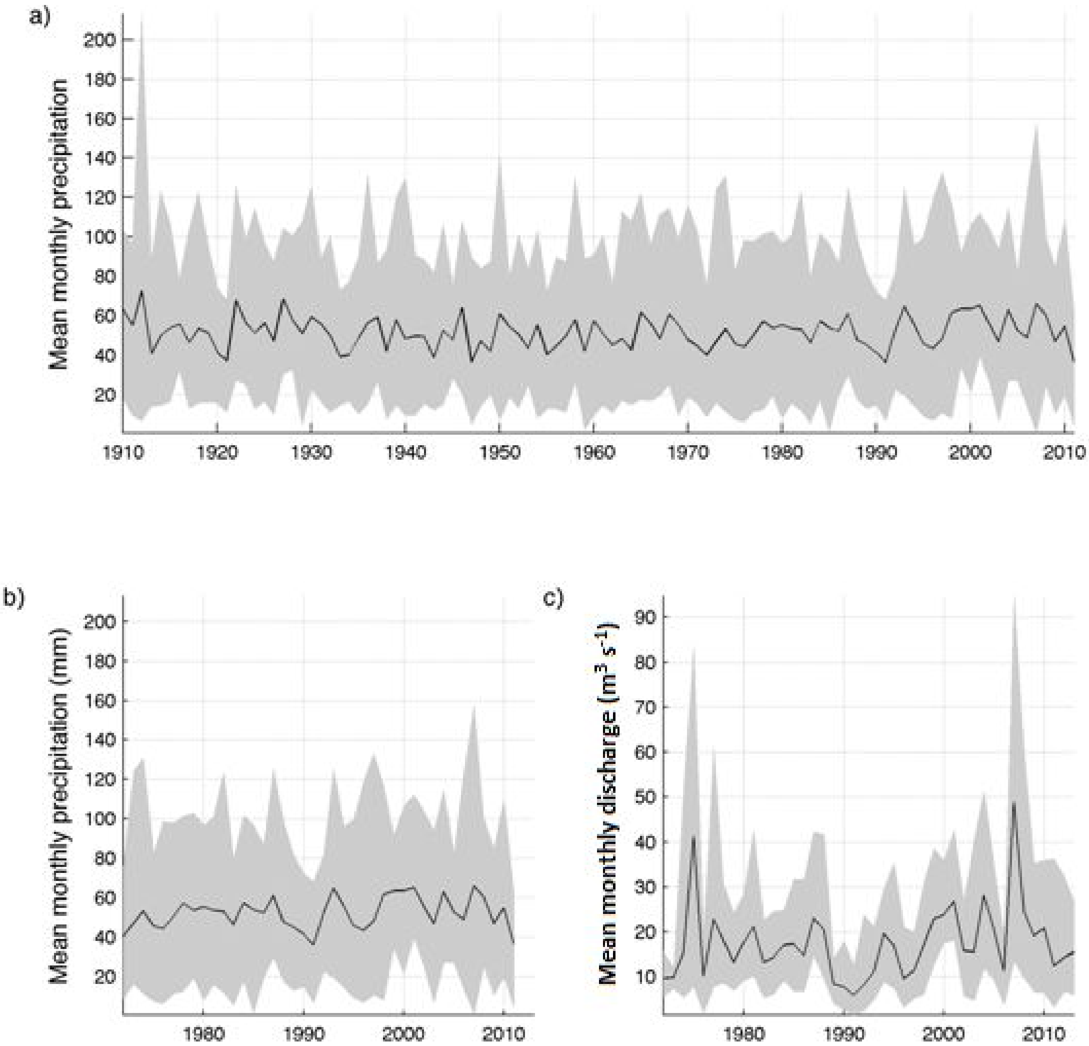



**Figure 3.5** Mean monthly precipitation time-series plot 1910-2011 showing the annual sum (a), mean monthly precipitation 1972-2011 (b), and associated mean monthly discharge time-series plot 1972-2011 (c) measured at the Environment Agency gauging station near Warham. Alteration of the flow regime was not associated with mean monthly precipitation.

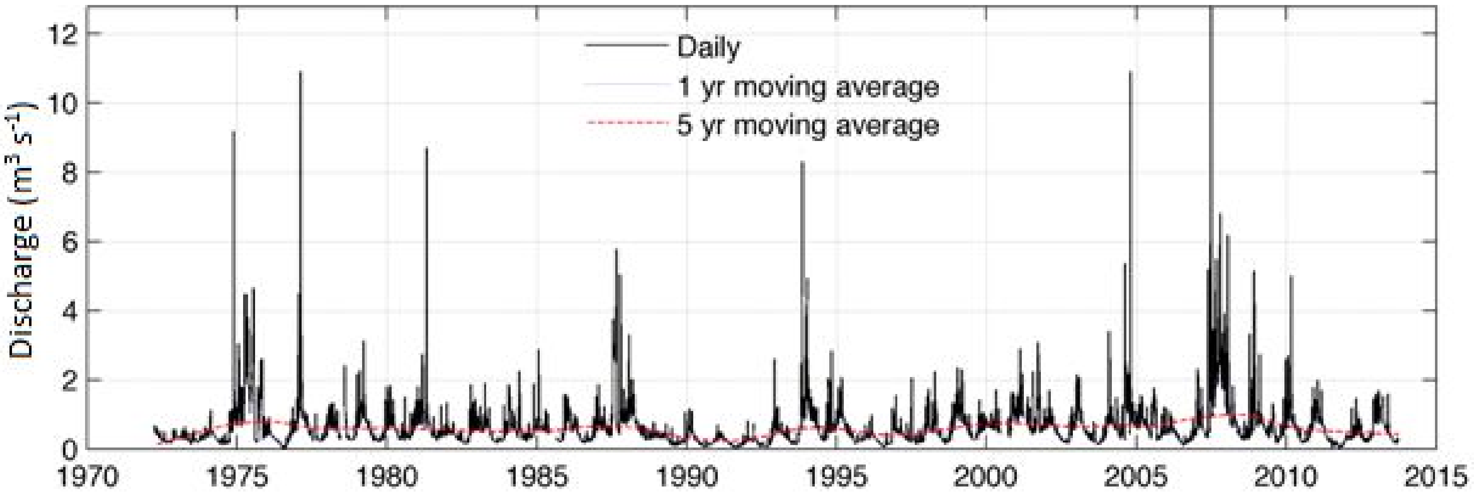



**Figure 3.6** Daily discharge time-series 1972-2013 with 1 year and 5 year moving averages, measured at the Environment Agency gauging station near the village of Warham. Discharge was characteristically low, punctuated by high annual discharge events. Long-term variability remained low.

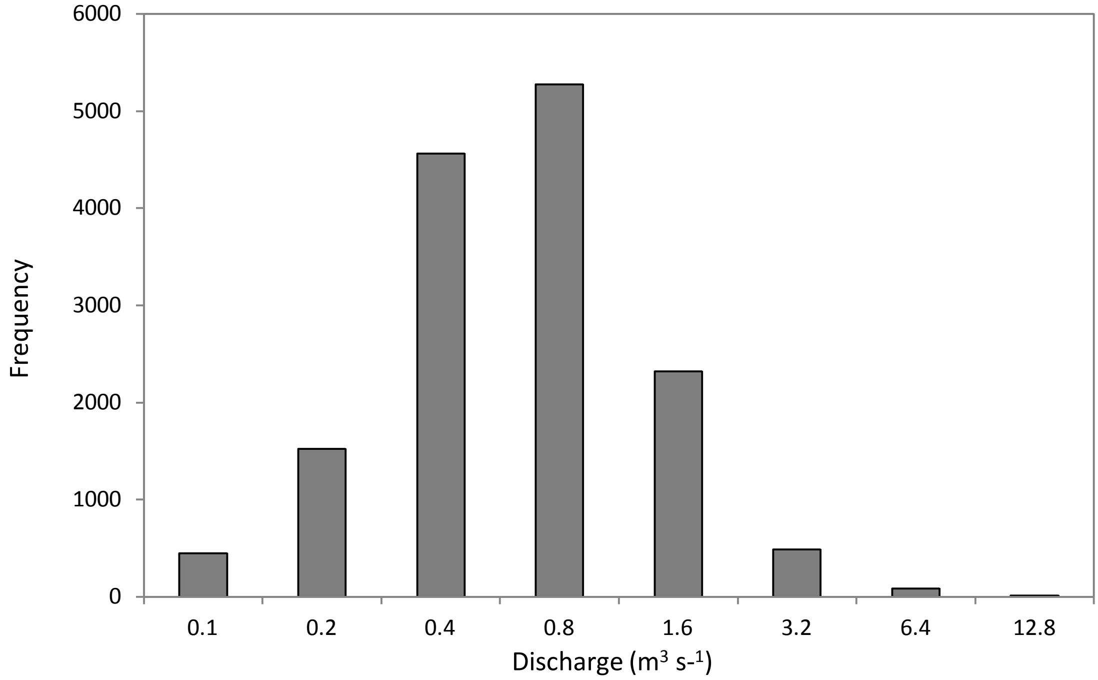



**Figure 3.7** Frequency histogram of mean daily gauged discharge ($m^3\ s^{-1}$) 1972-2013 measured at the Environment Agency gauging station near the village of Warham. Discharge was characteristically low, <0.8 $m^3\ s^{-1}$. Frequency of discharge ≥1.6$m^3\ s^{-1}$ became less common with increasing discharge. High discharge events, >3.2$m^3\ s^{-1}$ were rare over the time period 1972-2013.

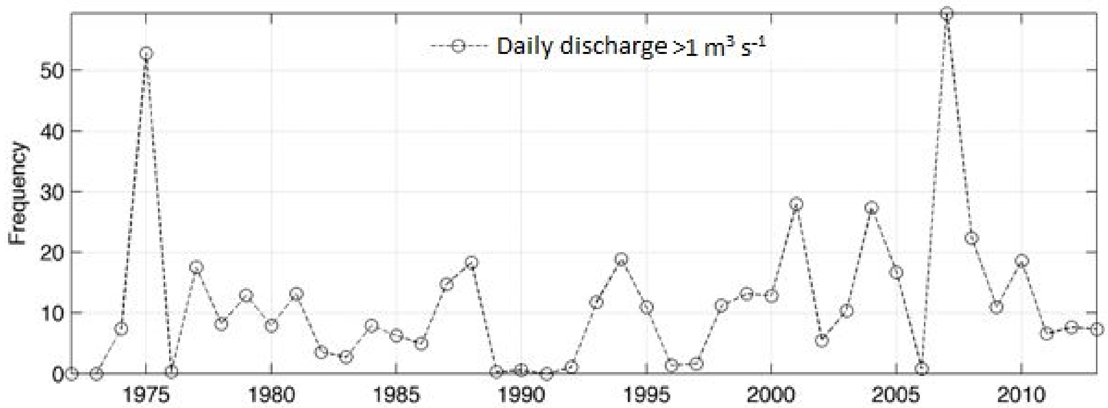



**Figure 3.8** Frequency of daily discharge >1$m^3\ s^{-1}$. Discharge >1$m^3\ s^{-1}$ was low overall throughout the time-series. The years 1975 and 2007 had over 50 days of discharge >1 $m^3\ s^{-1}$. However, most years had <20 days of discharge >1$m^3\ s^{-1}$.

The hydraulic regime was characterised by greater mean discharge during the winter months (0.78 $m^3 s^{-1}$), whilst mean summer flows were characteristically much lower (0.41 $m^3 s^{-1}$) reflecting the more consistent nature of winter rainfall recharge to the underlying chalk aquifer (and the low frequency of summer rain storm events) (Figure 3.9). Summer baseflows were maintained by late summer and winter precipitation recharge of the chalk aquifer. Discharge steadily increased from October to December and decreased from April to June as aquifer water levels dropped. High summer rainfall variation caused greater variation in summer discharge reflecting the increased flow response period to precipitation caused by the glacial geology of the upper reaches. However the annual cyclic nature of discharge is nested mostly within a narrow range of low discharge values. Very high discharge events were rare but had a greater likelihood of occurring during summer (June) or autumn (October and November) (Figure 3.9) reflecting past intense rainfall events during these months.

Stream competence, the maximum particle size (for e.g. sand, gravel, pebble), and capacity, a measure of the maximum solid load (bed and suspended) a stream can transport, is controlled by discharge (Reid et al., 1997). As discharge increases through a particular reach, shear stress and the associated ability to transport larger particle sizes increases (Hooke, 1975). Channel processes and stream competence in the River Stiffkey were controlled by characteristic low discharge; small clast sizes such as silt ($0.004<D\leq0.06$ mm) were more readily eroded and transported as the capacity to transport fine material was high (Figure 3.10). Less frequent discharges of mid-magnitudes were able to redistribute sands (0.3-0.55 $m^3 s^{-1}$) and gravels (0.55-1.5 $m^3 s^{-1}$) whilst the competence to redistribute pebbles (1.5-3 $m^3 s^{-1}$) and cobbles (>3 $m^3 s^{-1}$) occurred very infrequently, c. 1975 and again c. 2006. However, discharge between 1972-2011 had sufficient energy for gravel transport. Low discharge in the early 1990s had an associated low stream competence. The capacity for sand transport was however maintained by stream competence during this time. Such flow regimes are characteristic of streams with a high degree of gravel bed armouring.

The greatest input of agriculturally-derived sediment enters the river channel during late summer and autumn months. This is due to a combination of factors. Autumn sown crop varieties have dominated agricultural practice over the past 50 years. Fields during autumn are not vegetated and as such ensures a large supply of sediment available for erosion by heavy precipitation. The road and farm track network act as a conduit directing sediment-laden run-off towards the river channel (Figure 3.11).

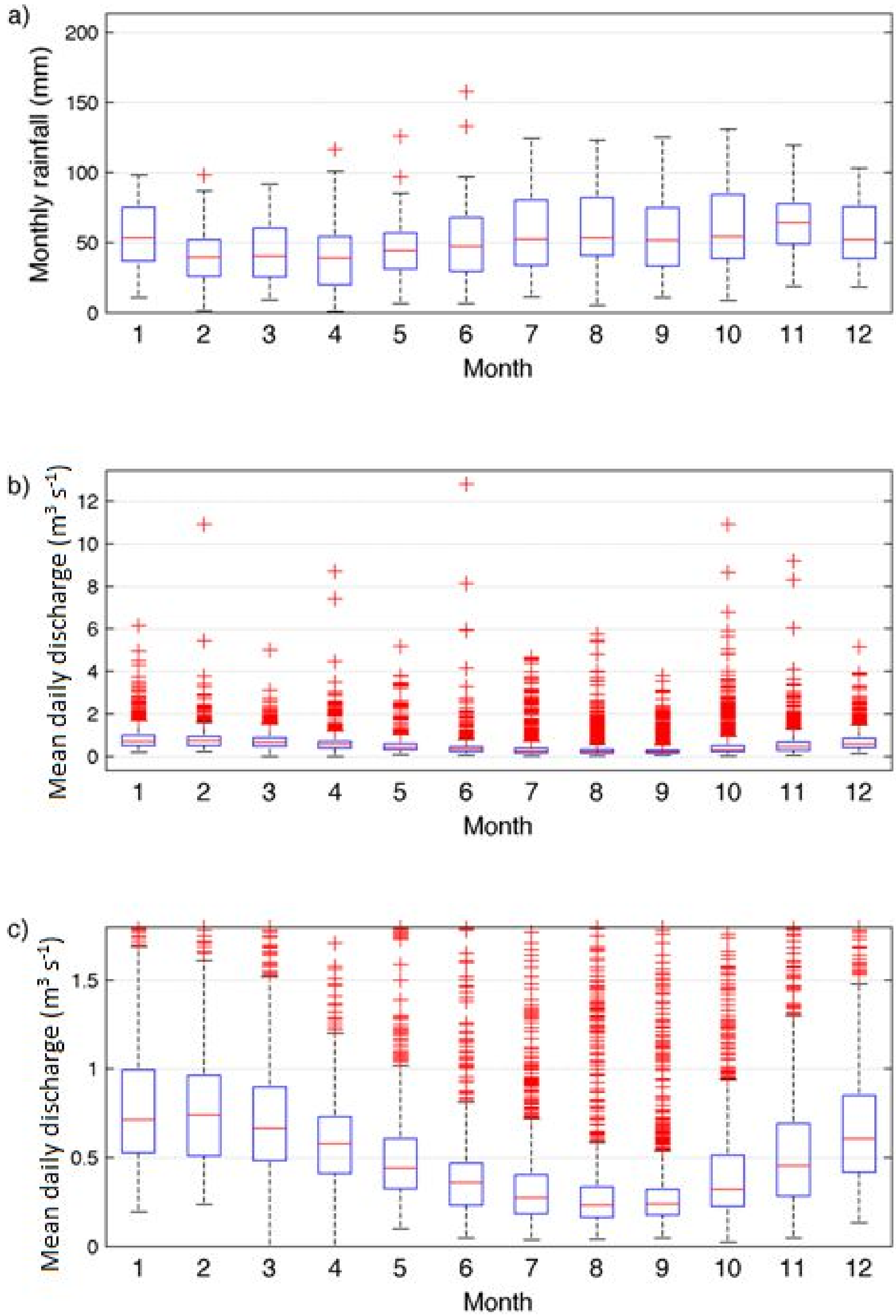



**Figure 3.9** Boxplot of monthly rainfall (a), and mean daily discharge plotted for each month over the period 1972-2011 (b). Extreme outliers of mean daily discharge were removed for expansion of y-axis of mean daily discharge on a monthly basis (c). Discharge was maintained by late summer and winter rainfall. However, variation in summer rainfall led to high variation in summer discharge reflecting the glacial geology of the upper reaches that increased the flow response period to precipitation.

Agriculturally-derived sediment is characteristically comprised of the more readily mobilised finer grained sediment fraction. Vast quantities are eroded from the steeper western divide, transported along the road network and deposited in the river channel at low lying points, often bridge or ford crossings, for example the Wighton village road bridge (Figure 3.11). This specific run-off pathway lies in an ephemeral dry valley that flows through the village of Wighton after heavy precipitation (see section 3.3.2), frequently transporting large quantities of sediment to the river channel.

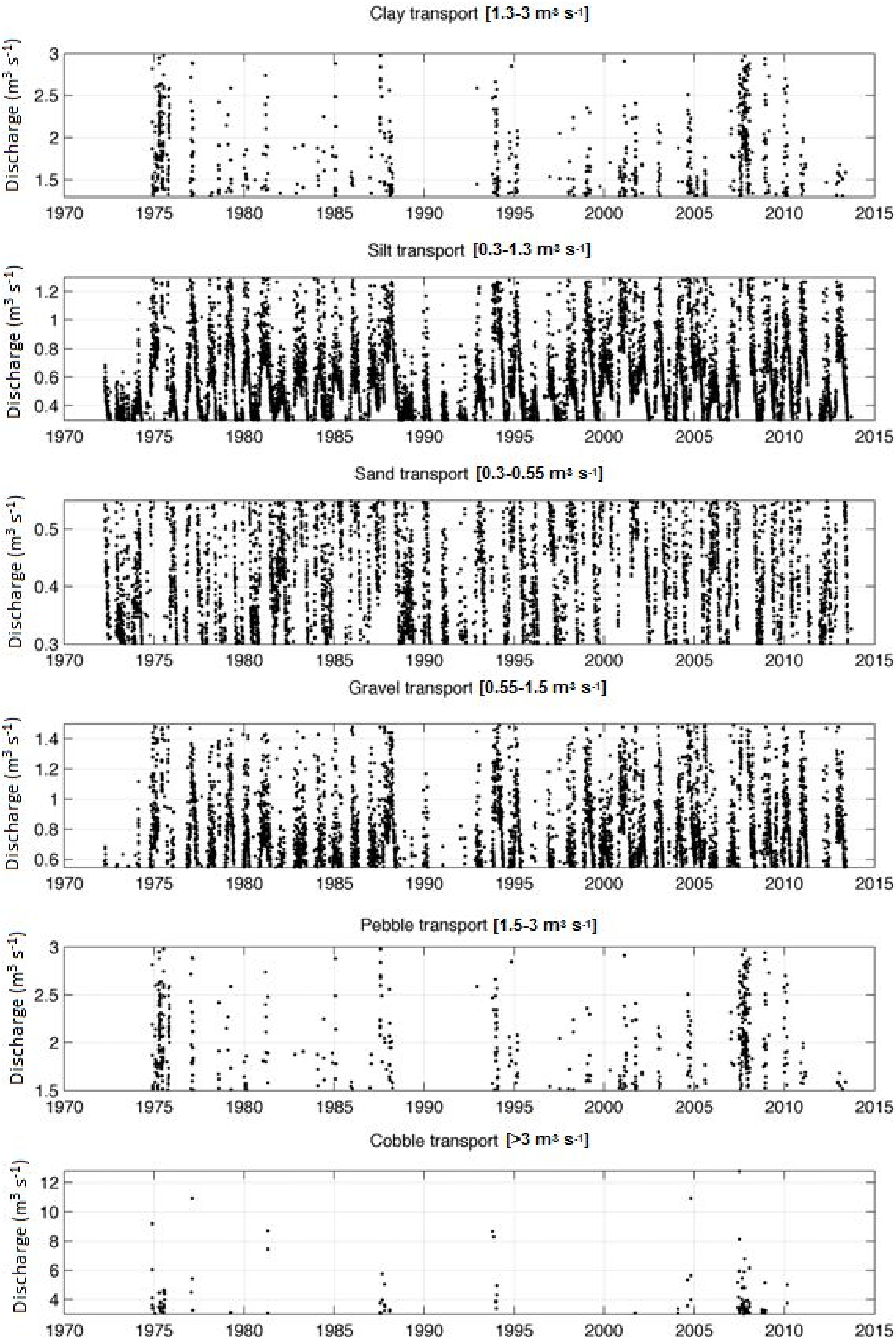



**Figure 3.10** Time-series scatter plot of discharge and its association with specific modes of sediment grain-size transport competencies. Sediment clast sizes were based on Wentworth (1922). Transport velocities were based on Hjulström (1935). Finer grained sediments were most readily transported.

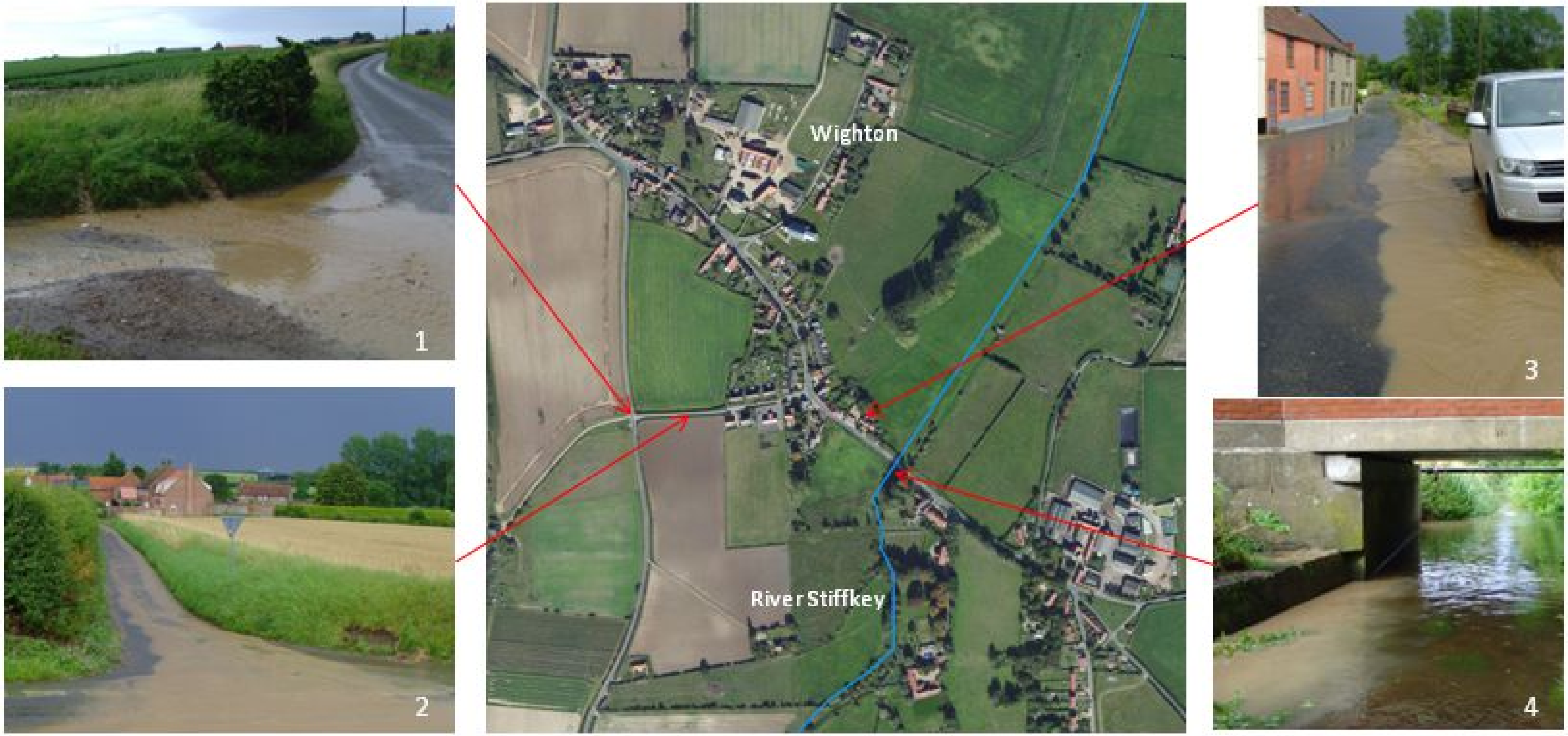



**Figure 3.11** Photographic evidence confirming a sediment-laden run-off event triggered by convective summer rainfall. Sediment, eroded from an arable field, was transported downhill as surface run-off (1), using road pathways and farm tracks as a conduit connecting fields to river channel (2) and (3), and discharged into the River Stiffkey at the lowest elevation where sediment pathways converge (4). Run-off followed an identified key flow pathway channel of an ephemeral dry palaeo-channel tributary, see section 3.3.2 below (Figure 3.15).

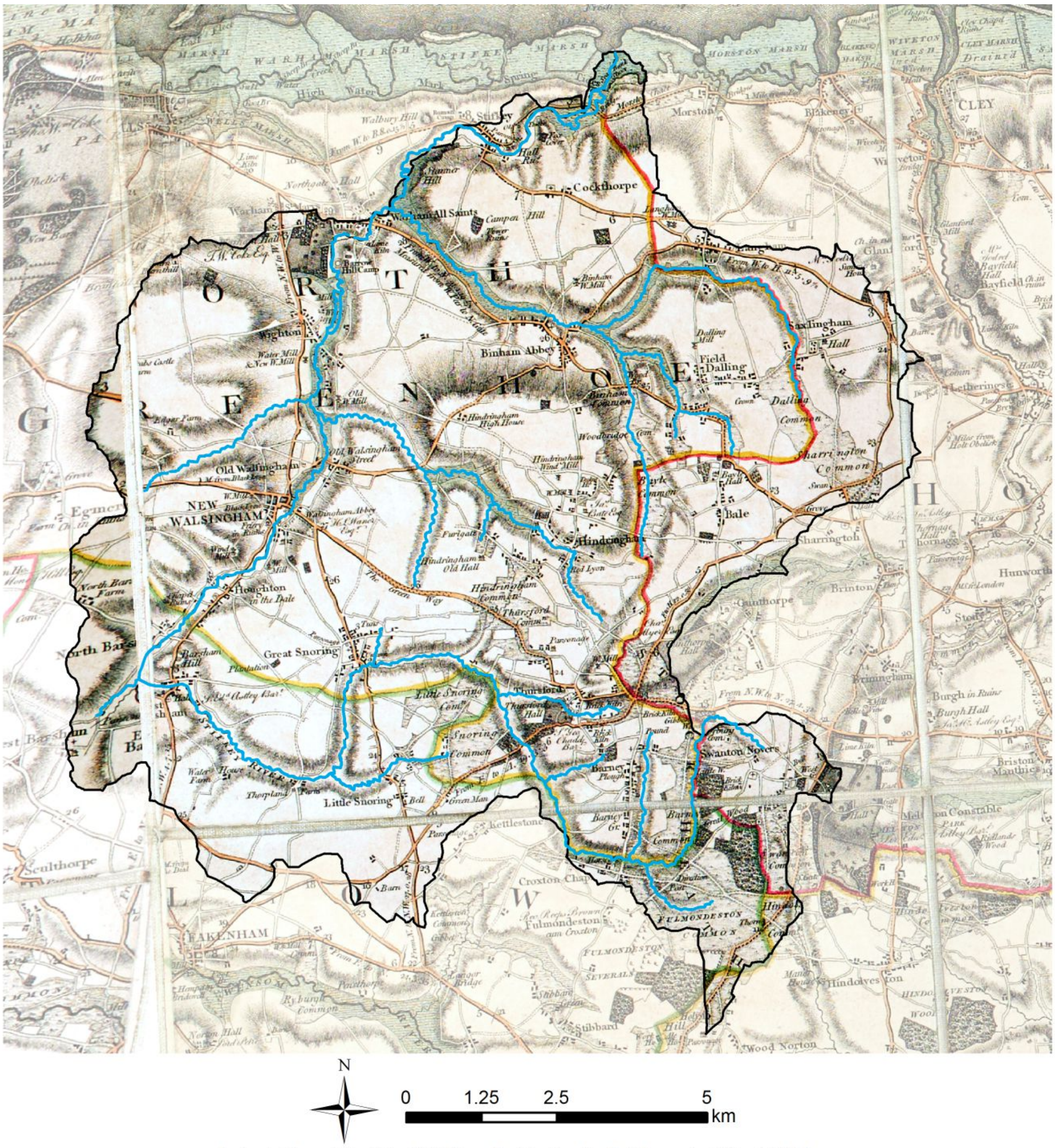
## 3.3 Catchment land-use and river channel modification: alteration of the sediment regime

### 3.3.1 Historical perspectives of land-use within the River Stiffkey catchment

Agricultural domination of the River Stiffkey catchment extends back to at least the late 18th century, observed by semi-regular field boundaries mapped by William Faden in 1797 (Figure 3.12). Field boundaries of the Landmark Information Group 1849-1899 Country Series map (1:10560) were consistently well defined (Figure 3.13), indicating an established agricultural economy throughout the catchment during the mid- to late 19 century. These boundaries were likely established through the Inclosure (or Enclosure) Acts of the 17th to 19th century in which use of common grazing and crop lands were legally restricted to landowners.

This reliance on agriculture continued; in the mid 1930's >90.0% of land-use was designated as arable annual and perennial crops including ploughed land (Figure 3.14, Table 3.1) (Land Cover Map 2007; Morton et al., 2011). However, likely due to land-use reclassification and further developments in mapping precision, the total contribution of arable land-use has not been consistent; 74.6% in 1990, 80.3% in 2000, and 71.7% in 2007. Throughout this period land-use was nonetheless dominated by agricultural practice. The dominance of agriculture predisposes the river channel to a greater risk of catchment-derived sediment deposition (see section 3.2.2, Figure 3.11). Furthermore, the long history of agricultural practice infers that the catchment has been a source of readily available sediment since at least the mid-18th century. However, elevated sediment transport to the river channel would have began with large scale catchment deforestation c. 4000 AD (Eaton, 1989). Although very little woodland has remained since deforestation c. 4000 AD (Sheail, 1988; Gregory and Davis, 1997), woodland coverage within the catchment area has steadily increased from 2.5% in 1931 to 6.7% 2007. A relative increase in the abundance of grassland (heather, improved, neutral and rough) was observed between 1930-2007, from 5.5% to <20%. These increases in grassland and woodland were likely either a response to an increased demand for greater environmentally sound farming practice, or a result of land-use reclassification between years 1931-2007.

Urban development has subsequently remained minimal (Figures 3.13 and 3.14). Indeed, North Norfolk is one of the least urbanised regions of England; in 2007, the most current land-use data set for North Norfolk, <1.5% of land-use in the River Stiffkey catchment was classed




Faden's Map of Norfolk, 1797. Supplied by the Norfolk Records Office (2015).

**Figure 3.12** The River Stiffkey catchment 1797 marked on Faden's map of Norfolk. This was the start of regular field boundaries in North Norfolk. Note the sinuous river channel, as well as how the mapped river line extends beyond the catchment boundary. This was likely due to inaccuracies caused by the cartography methods of the late 18th century.

as developed for urban/rural use (Figure 3.14, Table 3.1). The increase in 1990 to 5.8% was likely due to an alternative classification system derived from the Landsat 5 Thematic Mapper satellite imagery. The Land Cover Map (LCM) 2000 data were derived from satellite imagery but based land-use on the UK Biodiversity Action Plan (BAP) (Fuller et al., 2002). Land Cover Map (LCM) 2007 was an update of LCM 2000 (Morton et al., 2011).

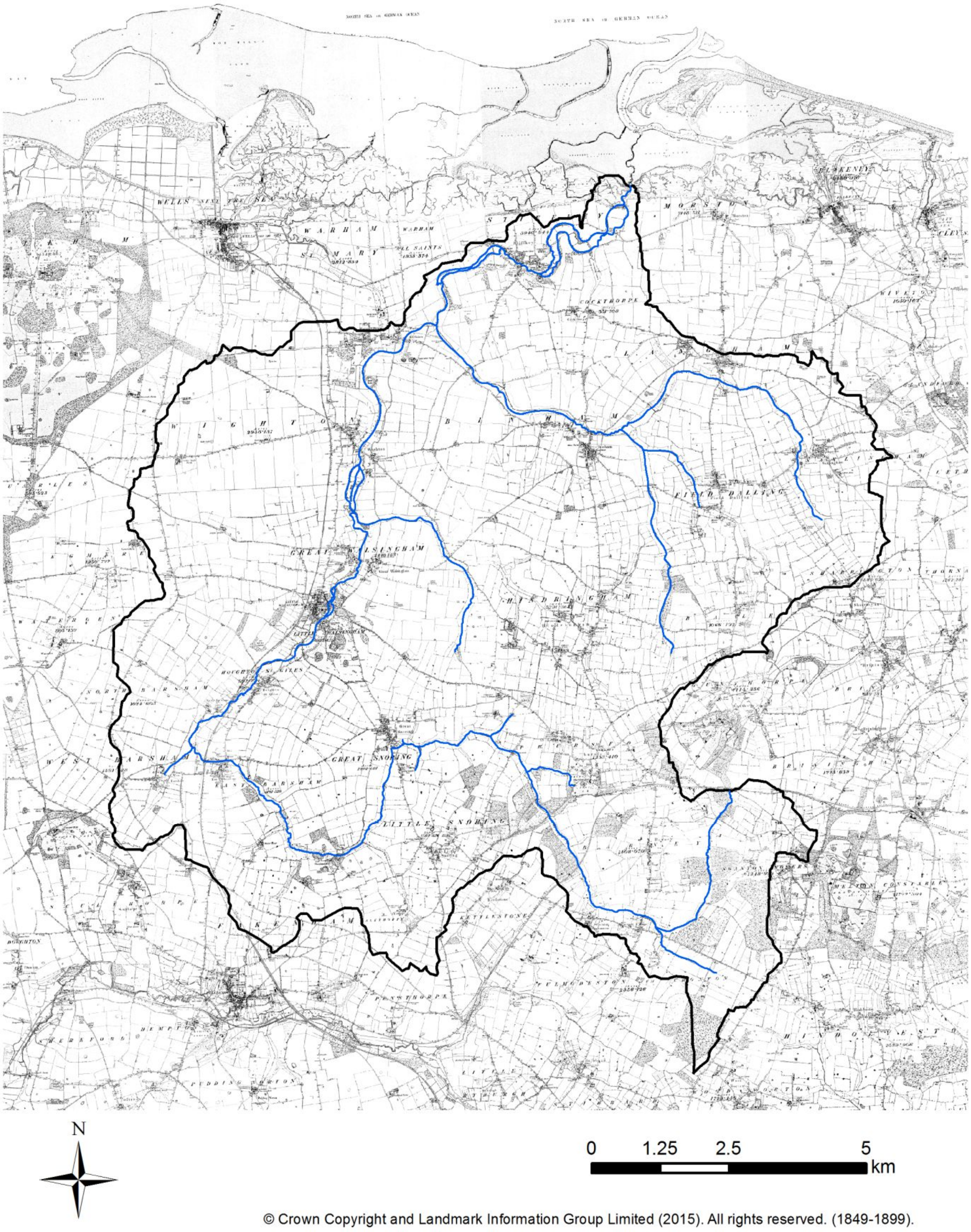



**Figure 3.13** Landmark Information Group 1849-1899 Country Series map (1:10560). Note the abundance of well marked and regular field boundaries, the result of Enclosure Acts of the 17th to 19th century that defined ownership of arable and grazing field limits.

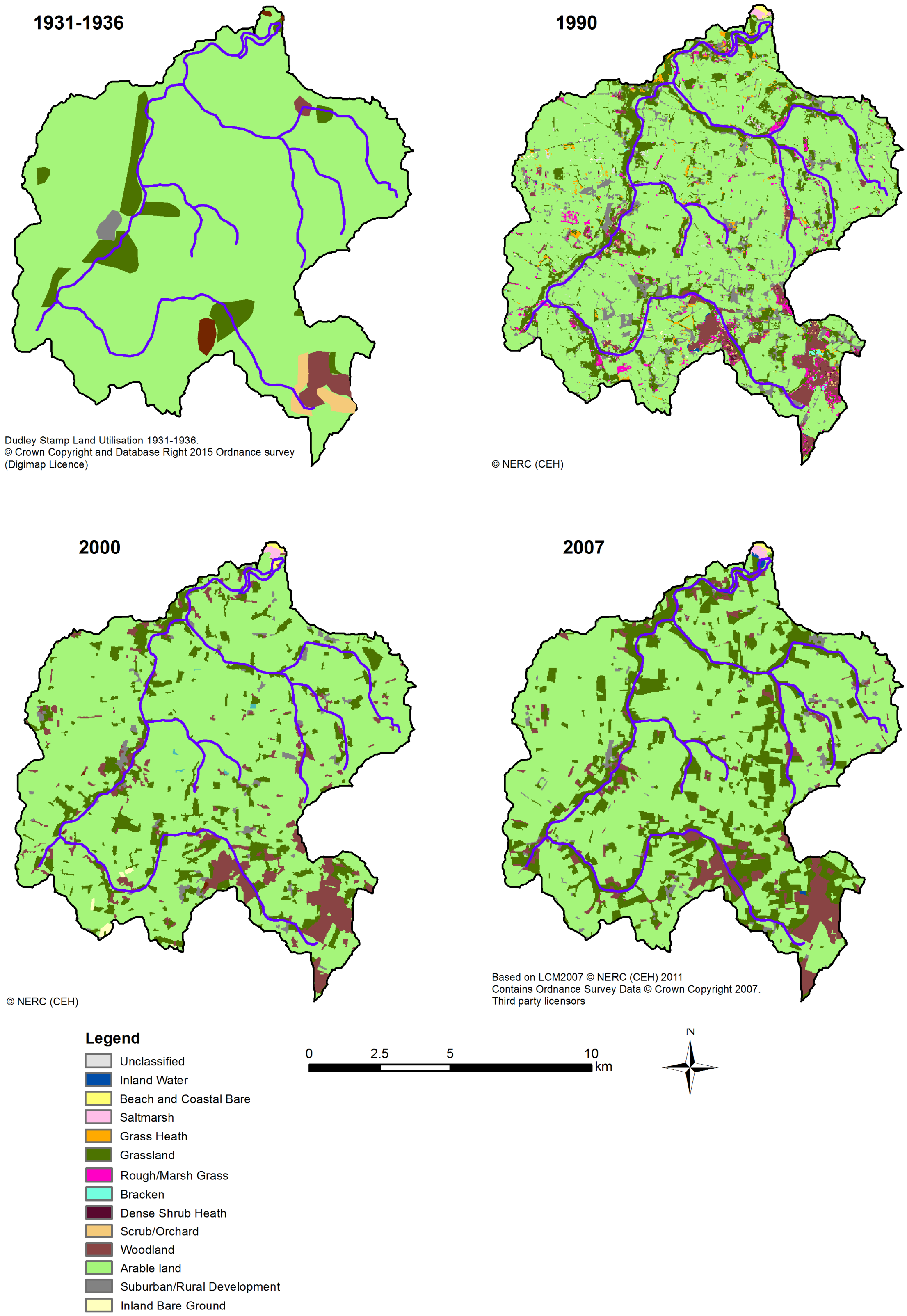



**Figure 3.14** Land-use in the River Stiffkey catchment from 1931-2007 reflecting agricultural dominance for the majority of the 20th century. The catchment has few remaining woodlands and grasslands, although an increasing trend was observed, see Table 3.1 below.

### 3.3.2 River channel modification and rehabilitation: implications for channel processes

With the exception of high stream discharge during 1975 and 2007, there was low variability over the time period 1972-2013, and as such significant modification of the river channel prior to the late 18th century would have had considerable impact on the flow regime of the river. River channel complexity has been anthropogenically simplified over time in response to increased agricultural and social pressures (Figures 3.15 and 3.16) (Boon, 1992; Hey, 1996; Wharton, 2000; Charlton, 2008).

Key flow pathways, a proxy for the pre-modified river channel, provided an indication of where water accumulated and flowed on the basis of landscape topographic features (Figure 3.15). Such topographically derived flow pathways were indicative of the perennial River Stiffkey channel as well as ephemeral surface waterways such as might be observed during heavy rainfall events. In the past these areas would have been a marsh-like environment bridging aquatic and terrestrial ecosystems (Sheail, 1988; Gregory and Davis, 1997). Three east-flowing ephemeral tributaries were observed in the lower reaches of the key flow pathways river channel, a-c (Figure 3.15). However, ground truthing confirmed that all three of these tributaries are now completely terrestrialised and constitute part of the agricultural land area of the catchment (see Woodridge and Goldring, 1953). These upper dry valleys, a common element of the chalk stream landscape, are palaeo-tributaries from a past environment characterised by elevated water tables (Berrie, 1992). Under conditions of reduced abstraction and high rainfall, these dry valley tributaries could begin to flow, if only temporarily (Berrie, 1992). It is not known whether these tributaries flowed under the rainfall conditions observed in 1975 and 2007. It is likely that a significant amount of rain would be required to induce flow. Tributary b) (Figure 3.15) does indeed respond to heavy rainfall (Figure 3.11). Although the dry valley flowpath direction has likely been altered somewhat through the development of roads and buildings in Wighton, which reduces drainage and increases run-off, a natural run-off response remains in this valley. Terrestrialization of these dry valleys is a good indication of early alteration of the river channel, likely through land drainage (Sheail, 1988; Park, 2005a; Clark, 2005; Watson, 2005). The northern-most tributary, marked a) on the key flow pathways map (Figure 3.15), was likely drained by subsurface clay pipes, known as arable tile drains. An outlet from a clay pipe was readily observed with a continual flow of water entering the main channel. It is likely that this area was drained as part of a farming scheme prior to the latter half of the 18$^{th}$ century. Field drains have been reported as a significant source of suspended sediment yields. In sediment source fingerprinting studies of agricultural catchments, field drains contributed >50% of the total suspended sediment yield (Russell et al., 2001; Palmer, 2012). The river channel underwent these modifications prior to the start of the 19th century, observed by comparing key flow pathways and the historic 1797 river channel traced from Faden's map of Norfolk, one of the earliest maps of the region (Figure 3.15). It is likely that the sinuous nature of the River Stiffkey river channel, observed in the late 18th century, underwent longitudinal simplification throughout the 19th century, as recorded for many rivers in the UK (Brookes et al., 1983; Eaton, 1989). The historic river channel c. 1797 had a greater number of tributaries than the contemporary river channel, being 7 km longer. >10 000 m of river channel were straightened for flood mitigation, mostly in the lower reaches, to increase water conveyance to the sea (Figure 3.16) (Brookes et al., 1983). Such modification further simplified the river network (Wharton, 2000; Charlton, 2008); reduced stream sinuosity from 0.8 to 0.7 as well as the overall river length by 1.4 km for those reaches where palaeo-channels were observed (Figure 3.16). Low resolution of the DTM at the scale required to identify palaeo-channels limited the determination of past stream width and depth variables. This was a likely indication of the once shallow character of the channel, typical of chalk streams (Berrie, 1992; Sear et al., 1999). Straightened channel reaches were also dredged in 1978 to increase channel depth and thus water conveyance (NRFA gauged daily flow dataset for Warham gauging station, 1972-2013) observed at the reach scale.

The structural design and location of rehabilitation gravels in modified river reaches changed the natural dynamic of channel processes at this scale (Figure 3.17). Neither the location in the river nor the physical structure of rehabilitation gravels replicate the natural gravel bed. Indeed these gravels alter natural channel processes at the macrohabitat and reach scale in the locations into which they were installed. Rehabilitation gravels were installed in three straightened and dredged river reaches (Figure 3.18). Reach 1 and 2 each had 3 rehabilitation gravels installed. Reach 3 was not dredged and was therefore shallower than the other reaches with a natural gravel bed. No rehabilitation gravels were installed there. Reach 4, characterised by a deep channel, had 7 rehabilitation gravels introduced. The isolated and ingot like nature of rehabilitation gravels were clearly distinguished by the longitudinal section of the river bed (Figures 3.17 and 3.18). Rehabilitation gravels reduced the slope angle of the surface water, observed as a flattening or levelling out of the water level. Furthermore, rehabilitation gravels were associated with holding up river water upstream, increasing the water level in order to pass over the gravel structures. Indeed, rehabilitation gravel 2009E from reach 2 was removed for this implication as it interfered with water level readings at the EA Warham gauging station.

**Table 3.1** Summary of the area (km2) and percentage land-use cover associated with Figure 3.14 above. The catchment is agriculturally dominated, >70% of current land-use. A trend of increasing woodland and grassland was also observed over the period 1931-2007. Not all land-use categories were consistently used between each land-use data set, likely due to reclassification, and as such there are missing values.

| | 1931-1936 | | 1990 | | 2000 | | 2007 | |
|---|---|---|---|---|---|---|---|---|
| **Land-use category** | **Area (km2)** | **Area (%)** | **Area (km2)** | **Area (%)** | **Area (km2)** | **Area (%)** | **Area (km2)** | **Area (%)** |
| Arable | 123.88 | 91.09 | 101.57 | 74.57 | 109.41 | 80.33 | 97.72 | 71.74 |
| Beach and Coastal Bare | - | - | 0.12 | 0.09 | - | - | - | - |
| Bracken | - | - | 0.34 | 0.25 | - | - | - | - |
| Dense Shrub Heath | 0.63 | 0.46 | 0.18 | 0.13 | 0.06 | 0.04 | - | - |
| Grass Heath | - | - | 1.28 | 0.94 | - | - | - | - |
| Grassland | 7.50 | 5.51 | 15.72 | 11.54 | 14.55 | 10.68 | 27.14 | 19.93 |
| Inland Bare Ground | - | - | 0.11 | 0.08 | 0.28 | 0.20 | - | - |
| Inland water | - | - | 0.09 | 0.06 | 0.01 | 0.01 | 0.16 | 0.11 |
| Littoral Sediment | - | - | - | - | 0.17 | 0.12 | 0.09 | 0.07 |
| Rough/Marsh Grass | - | - | 2.46 | 1.80 | - | - | - | - |
| Saltmarsh | - | - | 0.14 | 0.10 | 0.20 | 0.15 | 0.22 | 0.16 |
| Scrub/Orchard | - | - | 0.26 | 0.19 | - | - | - | - |
| Supra-littoral Sediment | - | - | - | - | 0.10 | 0.07 | 0.01 | 0.01 |
| Unclassified | - | - | 0.39 | 0.28 | - | - | 0.00 | 0.00 |
| urban/suburban/rural | 0.66 | 0.49 | 7.90 | 5.80 | 1.97 | 1.44 | 1.88 | 1.38 |
| Woodland | 3.34 | 2.45 | 5.67 | 4.16 | 9.47 | 6.95 | 8.99 | 6.60 |

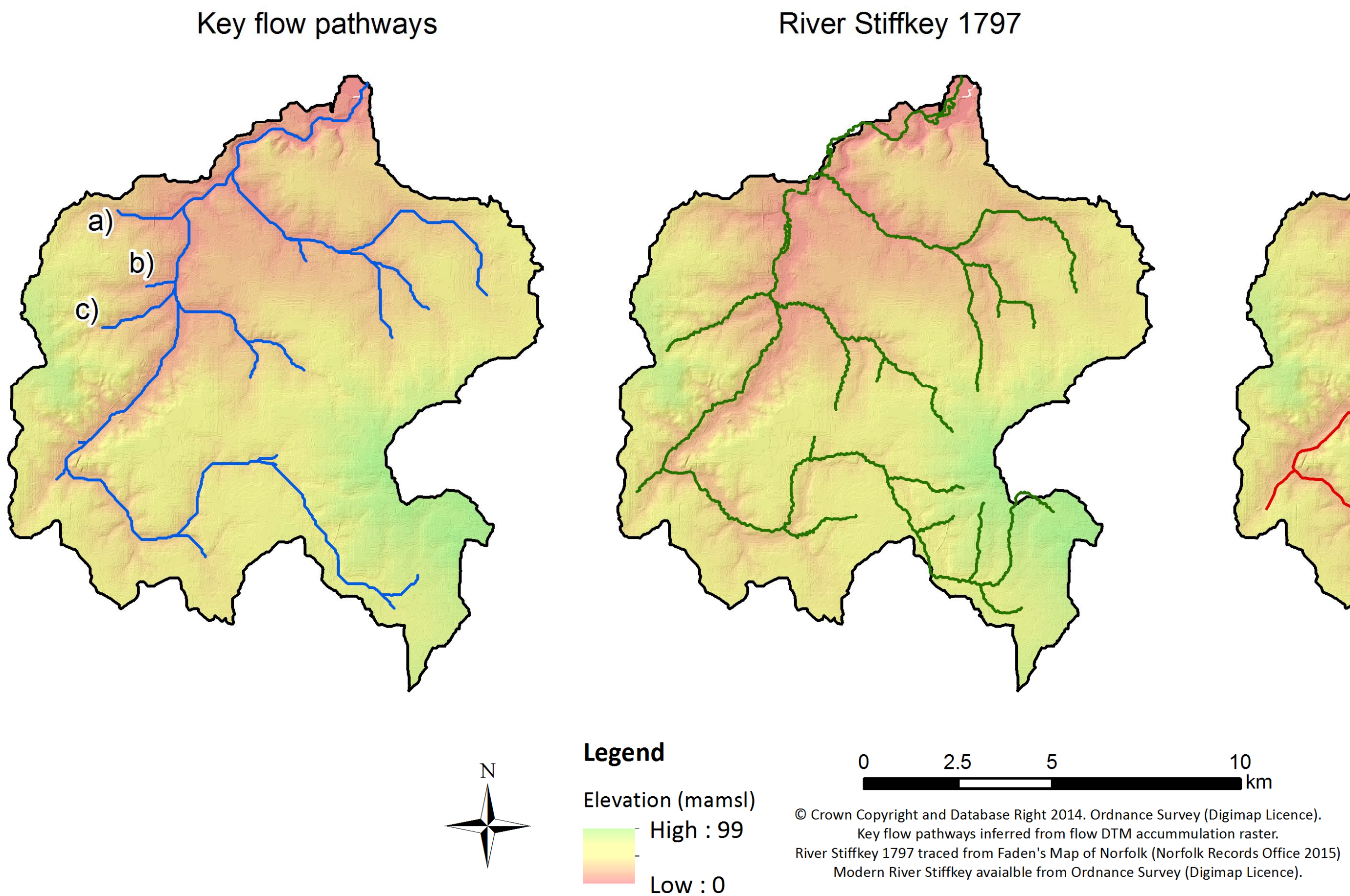



**Figure 3.15** Change in channel form over time. Key flow pathways, derived from topographic features, serve as a proxy for the pre-modified river channel. Land drainage and channel modifications in response to agricultural and social pressures had simplified the channel network by 1797, although it remained very sinuous. Further modification throughout the 19th and 20th century, largely for flood management, straightened the river channel to increase water conveyance to the sea, observed in the modern River Stiffkey channel planform.

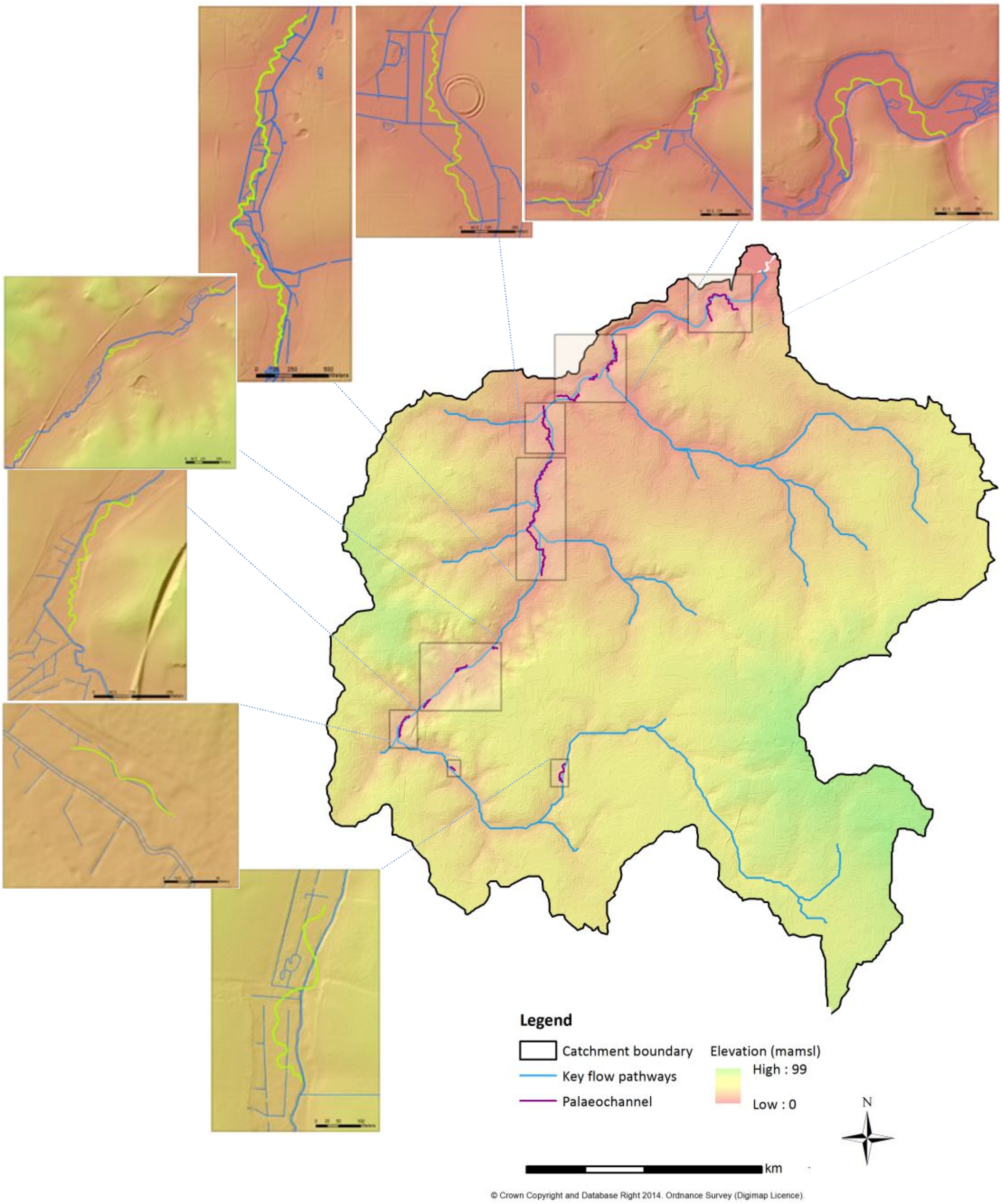





**Figure 3.16** Approximated palaeo-channels on the River Stiffkey main channel based on LiDAR DTM data. Reaches of the River Stiffkey channel have been straightened throughout most of its length. Significant engineering has altered natural channel dynamics through the mid- to lower river segments.

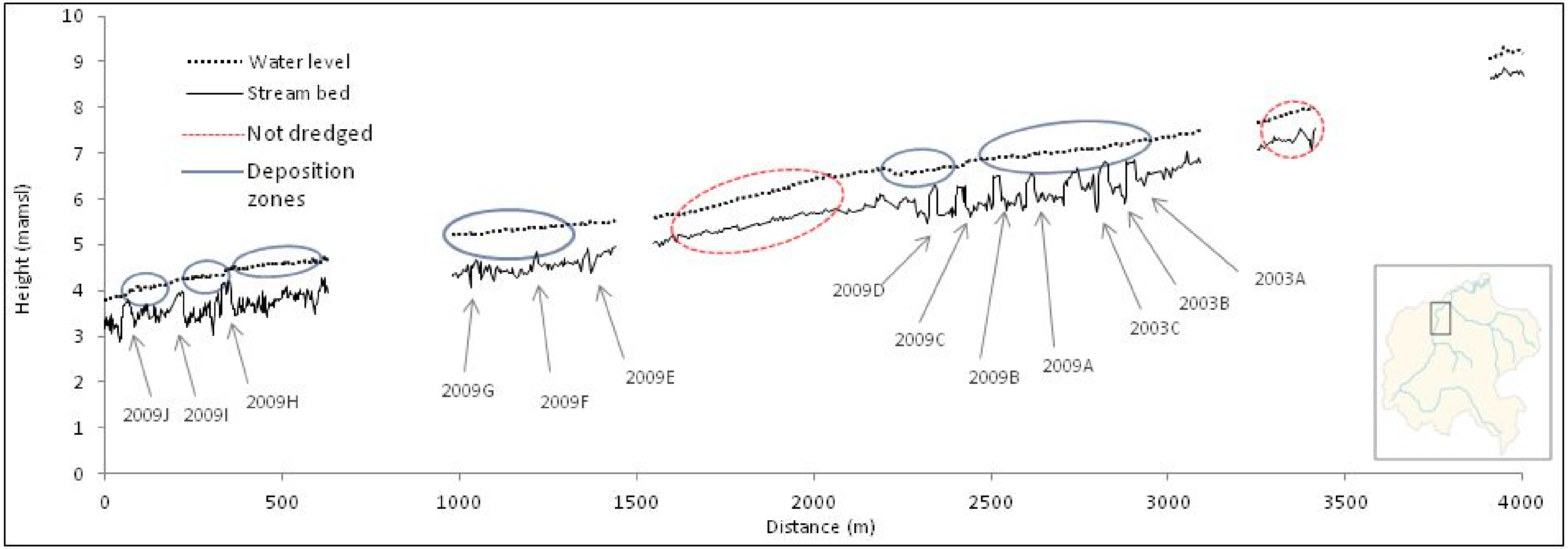



**Figure 3.17** The longitudinal channel profile indicating bed and water level measured in July 2012 using a dGPS set-up above an Environment Agency Bench Mark. Elevation data is AOD. Water surface level and river bed were measured simultaneously in an upstream manner at the same discharge. Dredged reaches of the study reach were distinguished by a decrease in the river bed. The rehabilitation gravel installed in these reaches had an elevated profile, decreased the water level slope angle and consequently localised stream velocity. Depositional processes therefore dominated these reaches. Gaps in the data were due to a loss in satellite signal and represent areas of stream channel where no data could be measured.

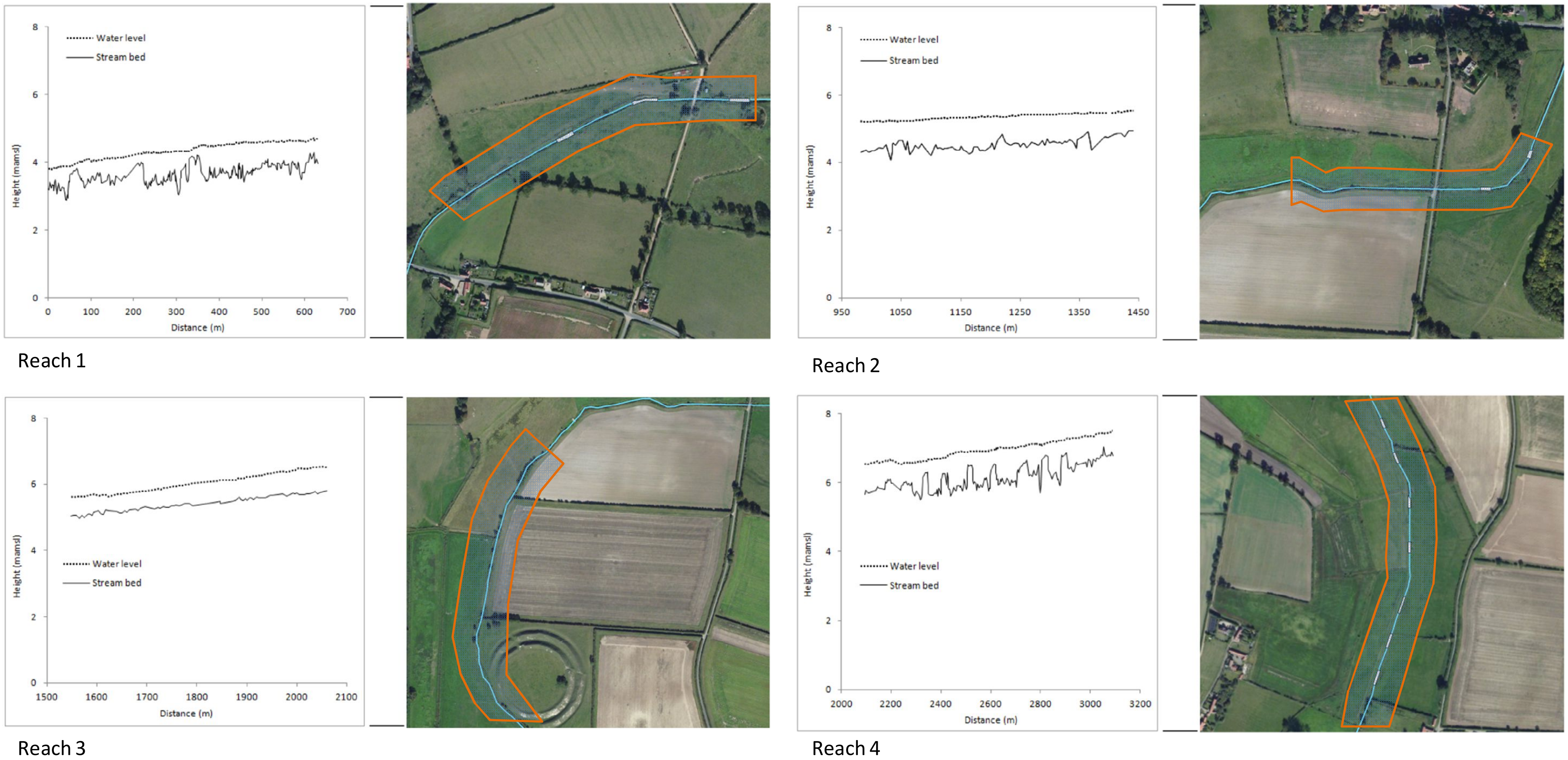



**Figure 3.18** River reaches indicating the three reaches where rehabilitation gravels were installed, reaches 1, 2 and 4, as well as a non-dredged reaches where no rehabilitation gravels were installed, reach 3. Source: Esri, DigitalGlobe, GeoEye, i-cubed, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community. Note how rehabilitation gravels formed peaked bedforms in the deeper dredged reaches.

In addition, the dredged reaches (1,2 and 4) had a much greater range of residuals around the mean (line of best fit) as a direct result of rehabilitation gravel introduction (Figure 3.19). This indicated the large difference between rehabilitation gravel crests and river bed, not observed on natural spawning gravel. Although there was no significant difference in residuals between each reach, there was a perceptible difference in standard deviations around the mean: 0.20, 0.12, 0.04 and 0.24 for reach 1-4 respectively. The natural gravel streambed in reach 3 had the narrowest range of residuals and the greatest water level slope inferring a higher stream capacity and competency. The loss of sinuosity, increased depth, decline in the slope angle of water surface and obstructions to flow will cause a decrease in velocity and as such sediment deposition will dominate local channel processes in these reaches. The single reach that had not been dredged or had rehabilitation gravels installed was shallower with an associated greater water level gradient. Inferred channel processes in this area would therefore exist in a state of greater equilibrium than in dredged reaches where rehabilitation gravels were installed.

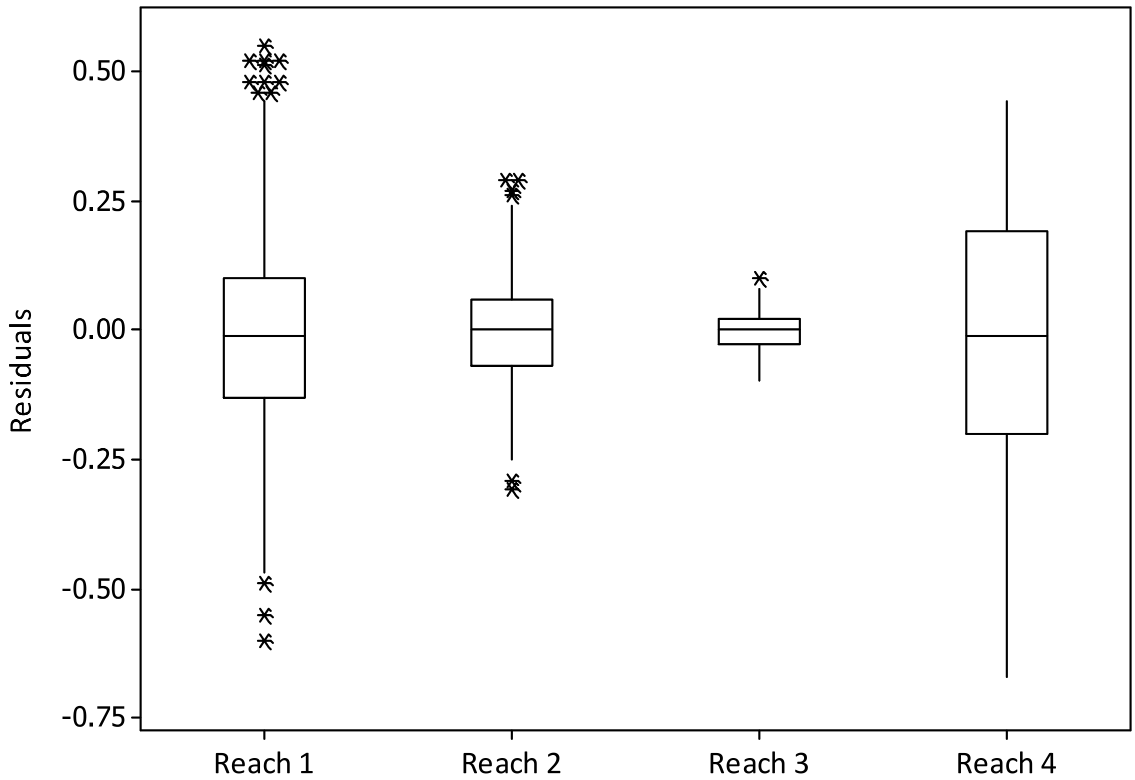



**Figure 3.19** Boxplot of linear regression residuals of each of the 4 reaches (Figure 3.18) associated with river bed gradient. Deeper dredged reaches where rehabilitation gravels were installed, reaches 1, 2 and 4, had a much greater range of residuals, whilst reach 3, consisting of natural streambed that was not dredged, had a much narrower range.

## 3.4 Discussion

### 3.4.1 Key catchment controls: implications for channel processes

Processes that determine river channel form: climate, topography, geology and vegetation operate at a large scale (Leopold et al., 1964; Hey, 1997; Wharton, 2000; Montgomery and Bolton, 2003), most notably, and useful for management, the catchment level. In this manner the catchment is the fundamental landscape unit for fluvial and sediment budget management (Everard, 2004). Understanding and recognition that channel-controlling processes operate at this scale are key to achieving sustainable river rehabilitation outcomes.

#### 3.4.1.1 Intensification of catchment land-use and sedimentation

Rivers are the key transfer mechanism of sediment from land to the sea. Anthropogenic impacts have artificially modified this dynamic altering the sediment flux by way of land clearance and agricultural land-use. The flux of sediment has been temporally variable throughout geologic time (Dearing and Jones, 2003). Climatic factors, and more recently anthropogenic influence, have altered the supply of sediment to river channels (Syvitski, 2003). Modifications of suspended sediment loads within the Holocene, however, are directly linked to anthropogenic activities, and not the direct result of climatic forcing (Peizhen et al., 2001; Dearing and Jones, 2003; Syvitski, 2003).

Modification of the River Stiffkey channel in response to land-use pressures have indeed altered the flow regime with no evidence of climatic forcing. Although anthropogenic activities have increased global fluvial sediment transport by 2.3±0.6 billion metric tons per year, 1.4±0.3 billion metric tons per year are being stored in rivers and not discharged into the oceans (Syvitski et al., 2005). Yang et al. (2002) and Wang et al. (2007) observed similar sediment retention on the Chinese Yangtze river and Huanghe (Yellow River) respectively. Undoubtedly the vast majority of this sediment is retained behind large reservoirs. Such modification to the sediment regime provides an indication of the impact anthropogenic activities have on global fluvial sediment flux. Forest clearance and the intensification of land-use (agriculture, logging, mining, urbanisation) have increased rates of soil erosion and consequently suspended sediment loads by orders of magnitude (Morgan, 1986; Dearing and Jones, 2003; Walling, 2006). There has been a 500% increase globally in land surface used for agriculture since 1700 (UNEP, 1995; Matson et al., 1997). This increased pressure has undoubtedly lead to sediment instability in river channels (Newson 1981).

Trustrum et al. (1999) reported increased sediment load inputs to the Waipaoa River, New Zealand following deforestation of the catchment in the mid 19th century (Murton, 1968). Intensive rainstorm events generated c. $15x10^6$ t suspended sediment load annually (Trustrum et al., 1999). In Britain significant deforestation between 6200-940 BP is inferred through an accelerated sedimentation rate of river channels (Dearing and Jones, 2003). Deforestation of East Anglian river catchments, approximately 5000 BP (Bennett, 1988; Eaton, 1989), exposed surface sediments to the mechanisms of water and wind transport thus making a large supply of sediment available for river channel processes. Cultivated land increases surface run-off through topsoil erosion and is associated with increased sediment load of river channels (Yang et al., 2002; Environment Agency, 2004). Walling (2006) indicated that it was possible for deforestation and cultivation to increase sediment loads over relatively short periods of time (c. 30-50 years).

Intensification of agricultural land-use post-World War II exacerbated the effects of deforestation and subsistence farming practice (Mainstone et al., 1999; Walling, 2006). Although farming practise intensified principally in response to population pressures (Walling, 2006), in the UK agricultural intensification was driven largely by the 1947 Agricultural Act which sought greater self-sufficient food production (Robinson and Sutherland, 2002). Greater reliance on mechanisation, autumn sown high-yield crop varieties, advances in chemical fertilizers and pesticides, scientific and technological advances and commercial land-use significantly increased crop yield (400%) during the post-World-War II period (Matson et al., 1997; Robinson and Sutherland, 2002; Walling, 2006).

Intensive contemporary agricultural practice has placed additional pressures on chalk streams. Chalk stream catchments are intensively farmed (Heywood and Walling, 2003; Walling et al., 2006, Jarvie et al., 2006) have high population pressures and low mean annual rainfall (Mainstone et al., 1999). Characteristically chalk stream catchments in England have a larger mean proportion of arable land-use (49%) than non-chalk dominated catchments (Environment Agency, 2004). Comparatively, arable land-use comprise >80% of the River Stiffkey catchment area, whilst woodland (6%) is >50% less than the average in England (Environment Agency, 2004). Woodlands have been indicated to significantly lower sediment flux from catchment to river channel (Matson et al., 1997). The Chinese policy of afforestation in the Yangtze River catchment in the late 20th century has successfully alleviated soil erosion caused by a history of deforestation and agriculture (Yang et al., 2002). Due to the paucity of woodlands, topsoil in the River Stiffkey catchment has a greater susceptibility to erosion and transportation.

In addition to sediment mobilisation through surface erosion, chalk streams are subject to further physical stressors, primarily abstraction. Given that their baseflows are groundwater dominated, chalk streams are becoming increasingly susceptible to over-abstraction. In fact Mainstone et al. (1999) purport that abstractions are the greatest threat to chalk stream hydrology. The trend of total water abstracted in England and Wales has decreased over the past 15 years (Department for Environment, Food and Rural Affairs, 2015). However, groundwater abstractions from the southern and eastern chalk aquifer of the UK remain high, accounting for 50% of total abstractions (Petts et al., 1999). Subsequently, total aquifer recharge in chalk catchments has been significantly reduced by agricultural intensification with consequential implications on flow regime (Mainstone et al., 1999). Further, groundwater abstractions can modify stream discharge, depth and substrate composition (Bickerton et al., 1993; Poff et al., 1997).

Abstracted low flows can be exacerbated by low winter recharge. Due to the proportion of flow delivered from groundwater sources in chalk streams, the reduction of stream velocity caused by abstractions are likely to influence suspended sediment dynamics (Heywood and Walling, 1999). Over-abstraction from the chalk aquifer has been shown to cause significant reductions in water levels, stream velocity and thus stream power affecting the ability to erode and transport sediments (Mainstone et al., 1999; Environment Agency, 2004). Bickerton et al. (1993) and Walling et al. (2006) observed increased fine sediment (<2 mm) deposition during periods of reduced (abstracted) flows on chalk streams. The loss of stream power caused by significant over abstraction in the River Glen in Lincolnshire, UK, has resulted in excessive fine sediment deposition with consequential implications for intragravel stages of *S. trutta* (Milan and Petts, 1998). Abstractions therefore impact on the physical stream character by shifting the dominant channel processes towards a depositional environment (Strevens, 1999; Petts et al., 1999). Sediment deposition caused by over-abstracting chalk catchments is a significant contributing factor for ecological decline (Wright and Berrie, 1987), primarily through the reduction of habitat diversity (Bickerton et al., 1993; Castella et al., 1995). Over-abstraction alters channel morphology and water quality (Armitage and Petts, 1992; Webb et al., 2003), invertebrate (Statzner et al., 1988) and macrophyte assemblages (Hupp and Osterkamp, 1996; Franklin et al., 2008) as well as fish migrations (Strevens, 1999; Environment Agency, 2004).

Although there is only a small rural population, significant agricultural pressures have placed a high demand on water use within the River Stiffkey catchment. Based on Environment Agency data (2013) abstraction from the River Stiffkey catchment is not sustainable and is causing an unacceptable environmental impact at low water levels. Wood and Petts (1994) as well as Wright and Berrie (1987), however, have indicated that strong ecological recovery from sustained low flows are possible, indeed for some biota it can be rapid (months) once normal flows are returned. The relationship between stream flow and sediment transport dynamics are relatively poorly defined (Wilcock et al., 1996a; Walling et al., 2006). However, stable channel morphology is maintained by a range of flows of which flooding is key to the maintenance of channel processes and consequentially habitat diversity (Reiser et al., 1989; Wilcock et al., 1996b).

#### 3.4.1.2 Hydraulic regime

Channel bed stability is determined by the hydraulic regime, channel morphology, stream gradient and the supply of sediment (Werritty, 1997). Like other chalk streams (Berrie, 1992; Mann et al., 1989), the River Stiffkey has a gentle catchment topography, is characterised by low stream discharge (<1m $s^{-3}$) and consequently poor potential for geomorphic activity, a characteristic chalk stream feature (Mainstone et al., 1999). The typical range of velocities (0.1-1.0 m $s^{-1}$) associated with chalk streams makes them susceptible to fine sediment accumulation (Berrie, 1992; Acornley and Sear, 1999; Petts et al., 1999), particularly in agriculturally dominated catchments (Mainstone et al., 1999). Sustained periods of low velocities encourages the development of streambed armouring (Reid et al., 1997). Surface armouring prevents finer particles beneath from being entrained during high flows and as such a greater velocity is required to remove the armour layer once it is formed (Charlton, 2008). A hydraulic regime dominated by low annual flows exacerbated by high levels of groundwater abstraction is characterised by an increased susceptibility of spawning gravel siltation. Low flows result in a poor frequency of in-channel sediment storage features such as gravel riffle habitat (Mainstone et al., 1999; Sear et al., 1999). Rehabilitation by means of gravel introduction is therefore only a suitable management strategy for chalk streams should the determinant sediment dynamics persist in a stable state. Similarly, any form-led gravel rehabilitation technique, such as gravel raking or high pressure gravel jetting, will require further maintenance as the constraints to ecological recovery will persist.

Geological characteristics play a key role in the nature of channel discharge and the hydrological response to precipitation events; impermeable geology has little retention time resulting in greater channel peak flows, whilst permeable geology moderates channel flow maintaining base flows during the dry season (Sear et al., 1999). Water has long (>20 years) residence times in the Chalk aquifer (Foster et al., 1986), providing the characteristic stable hydrological regime and regular annual discharge pattern associated with chalk streams (Mann et al., 1989; Berrie, 1992; Mainstone et al., 1999).

The main rivers of the North Norfolk region, Rivers Burn, Glaven and Stiffkey, are all south to north flowing and have high groundwater dominated baseflows; the River Stiffkey 76%, Glaven 85% and the Burn 95% (Environment Agency, 2005; Environment Agency, 2013). Due to the groundwater dominated hydraulic regime, these rivers all have UK BAP conservation designations, and the Stiffkey Valley is a Site of Special Scientific Interest (SSSI) (Oddy, 2014). Surface water run-off typically plays a very minor role in chalk stream hydrology (Mainstone et al., 1999; Berrie, 1992) as the permeability of the underlying geology attenuates the effect of rainfall on stream flow. However, many chalk stream catchments, particularly in East Anglia, have an impermeable geology constituent that increases the channel flow in response to rainfall (Mainstone et al., 1999). Recharge to the chalk aquifer is controlled in the upper reaches by overlaying glacial deposits. These clay-rich Chalky-Boulder Clay and sandy North Sea deposits (Hiscock, 1993) have a key role in controlling aquifer recharge and groundwater flow in the River Stiffkey catchment. Due to glacial deposits in the east of the region the groundwater regime of North Norfolk deviates from a typical chalk stream as the response time to precipitation is reduced (Hiscock, 1993). These deposits expose the River Stiffkey to a greater susceptibility of flash flooding (Environment Agency, 2005 and 2009). In this manner several North Norfolk rivers share similar hydrological regimes with limestone dominated catchments, although the mechanisms differ; characteristic fractures and fissures of limestone aquifers have shorter water residence times than chalk and therefore respond more readily to precipitation (Sear et al., 1999). The upper reaches of the River Stiffkey flow through glacial deposits and have a somewhat uncharacteristic chalk stream hydrologic response to precipitation. The River Burn in the west of the North Norfolk region has little or no glacial deposits and therefore has much greater (95%) baseflow originating from groundwater (EA, 2004).

Climate is a primary driver of sediment flux from land surface to river channel. Globally, approximately 50-75 Gt year$^{-1}$ of sediment are eroded from the land surface by rainfall

(Walling, 2006). Wang et al. (2007) associated a climate change induced decrease in precipitation with a 30% reduction in sediment load of the Huanghe River, China. Similar climate induced sediment transportation processes operate in the River Stiffkey catchment. Seasonal high magnitude rainfall events, primarily throughout the summer and autumn months, mobilise sediment from the dominantly arable catchment and, using the road and farm track network as a conduit, transport sediment-laden run-off to the river channel at road crossing points. Although chalk streams typically have little surface derived flow, development of the hard impermeable surfaces of urbanisation, road networks and informal farm tracks have artificially altered hydrological regimes, and consequently the sediment loads, of chalk streams (Mainstone et al., 1999; Walling and Amos, 1999). These impermeable surfaces facilitate sediment-laden run-off from arable farm lands to river channels (Walling and Amos, 1999). The widely accepted change in farming practice from spring to autumn sown crop varieties has increased susceptibility of fields to erosion during periods of high rainfall (Mainstone et al., 1999). Morgan (1986) as well as Dearing and Jones (2003) have argued that low order, small catchment areas have less buffering capacity and are therefore more susceptible to sediment erosion than larger catchments areas.

The capability of roads to act as a sediment conduit in Norfolk river catchments has been investigated by others (APEM, 2010; Evans, 2011; Collins, et al., 2013; Natural England, 2013). Natural England (2013) examined diffuse water pollution from road crossing points in the River Stiffkey catchment. Contrary to observations made during this study, the Wighton village road bridge was indeed a source for sediment pollution, whilst the arable nature of the catchment was the dominant underlying cause and not road degradation. No details or indication of the mechanism of sediment erosion or transportation were examined. Natural England (2013) submitted that the gentle catchment gradient would exclude direct field run-off, inferring an alternative source of sediment pollution in the River Stiffkey catchment. However, an investigation into fine-grained sediment (<2 mm) erosion and deposition on two chalk streams, the Rivers Pang and Lambourn in Berkshire, UK, conducted by Walling et al. (2006) demonstrated that slope was not a primary driver of diffuse water pollution, but instead concluded that land-use was. Cultivated fields generated significantly greater sediment through erosion than those used for pasture (Walling et al., 2006). Moreover, Walling et al. (2006) noted that the land surface was the dominant source of fine sediment deposition in river networks, whilst in-channel sedimentation processes had minor roles in the sediment budget. In-channel fine sediment storage of the mean annual sediment yield was <40% (Pang) and >20% (Lambourn). Very little (1%) deposited fine sediment eroded from within the catchment was discharged into the sea, indicating long residence times within the stream channel (Walling et al., 2006). Field drains are another important source of suspended sediment in agricultural catchments. Both Russell et al. (2001) and Palmer (2012) reported that field drains contributed >50% of the total suspended sediment yield in lowland agriculturally dominated catchments though sediment source fingerprinting studies.

Suspended fine sediment loads have been associated with high magnitude rainfall events in chalk stream catchments other than the River Stiffkey. Heywood and Walling (2003) associated elevated suspended fine sediment loads on the Hampshire Avon with high magnitude rainfall events. In comparison with the River Stiffkey, 80% of the Avon catchment is agricultural, however, mean winter rainfall is greater than summer, 76% and 59% respectively. Heywood and Walling (2003) observed that the reduced groundwater baseflow during summer further increased suspended sediment concentrations. Further, in their investigation of sediment dynamics of the upper River Piddle in Dorset, UK, Walling and Amos (1999) ascribed in-channel sediment accumulation to winter rainstorm mobilisation of the arable catchment. Farm tracks were used as conduits of sedimentation in a similar manner observed in the River Stiffkey catchment (Walling and Amos, 1999).

### 3.4.2 River modification: implications for reach-scale channel sedimentation processes

Channel processes are controlled by the hydraulic regime and sediment supply (Leopold et al., 1964; Gregory, 1992; Hey, 1997). Significant physical change at the catchment level, specifically land-use, vegetative cover and river channel modification, have impacted River Stiffkey sedimentation processes that are explicit at the reach scale. The low stream power, common on most chalk streams, has a narrow range of velocities capable of mobilising bed substrata (Mann et al., 1989; Acornley and Sear, 1999; Sear et al., 1999). Sediment supply within the River Stiffkey catchment is spatially and temporally variable determined largely by the dominant land-use, frequency and magnitude of precipitation and the associated hydrograph, topography, and the nature of the superficial geology. A change in fine sediment supply may force the channel to become unstable and undergo change (Werritty, 1997). An anthropogenically enhanced sediment supply to the River Stiffkey channel combined with the reduced stream capacity has led not only to system instability but an inability for natural geomorphic readjustment (Newson 1981; Werritty, 1997). Rehabilitation success of habitat heterogeneity schemes delivered exclusively at the reach scale in the River Stiffkey are therefore determined by (modified) catchment scale processes.

Chalk stream ecosystems are susceptible to anthropogenic impacts; just 37% of chalk streams in the UK are in a healthy ecological state (Environment Agency, 2004). Most, if not all, chalk streams in the UK are heavily modified or re-engineered to some degree (Gregory, 1997; Gregory and Davis, 1997). Alteration of river channels, such as deepening, straightening and widening simplifies the hydraulic regime and thereby complex hydrogeomorphic processes with consequent effects on sedimentation dynamics and stream capacity (Brookes, 1986; Petersen et al., 1992; Gregory and Davis, 1997). River channelisation modifies the hydraulic regime and causes energy disequilibrium. This induces a morphologic self-adjustment feedback mechanism towards a stable state through erosion and deposition processes, with consequent changes to sediment transport capacity (Brookes, 1985; Fryirs and Brierley, 2012; Landemaine et al., 2015). The period of self-adjustment that follows channelisation can have serious morphosedimentary implications (Brookes, 1985 and 1986; Landemaine et al., 2015). Channelisation has been identified as a significant factor responsible for the morphological degradation of river channels and is a principal cause of habitat diversity decline.

Decades of modification to the River Stiffkey have reduced the naturally high sinuosity associated with chalk streams (Sear et al., 1999), particularly in the mid to lower reaches. Channel straightening, as observed in the River Stiffkey, will initially increase stream velocity and therefore the sediment transport capacity, encouraging erosion and sediment load delivery to downstream reaches over the short term. Continual degradation will however result in slope decline and a predominance of depositional processes (Fryirs and Brierley, 2012). In this manner channel straightening reduces physical diversity, particularly the loss of pool-riffle morphology (Brookes and Gregory, 1983; Eaton, 1989). Artificially increasing channel width reduces depth and stream velocity and subsequently stream power and capacity (Mainstone et al., 1999; Fryirs and Brierley, 2012). Channel deepening is a leading cause of upstream channel incision, delivering sediment loads to the downstream reaches (Fryirs and Brierley, 2012). Moreover, deepening removes gravel beds. Mainstone et al. (1999) argued that the deficiency of geomorphic activity characteristic of chalk streams makes these gravels in effect irreplaceable. The River Stiffkey has been appreciably deepened and straightened through the mid to lower reaches (Figure 3.17). Sediment-laden run-off is stored at the reach scale in response to localised reductions in stream velocity as a consequence of channel modification within those reaches. The impacts on channel processes as a result of channelization are compounded by agricultural abstractions.

Landemaine et al. (2015) studied the sedimentation response to channelisation on the Ligoire, a French river. The Ligoire river catchment (82 $km^2$) is similar to the River Stiffkey catchment; it is largely agricultural (>70%), the underlying geology is dominated by Cretaceous chalk outcrops and has a gentle sloping valley gradient. The main channel (21 km) was completely straightened and re-sectioned in 1970, significantly altering stream morphology; 10% loss of channel length and bankfull width increased >60% with an overall loss of surface roughness. Immediately after channelisation water conveyance increased >300% and stream power rose by 80%. However, morphological feedback processes formed reaches dominated by deposition. Although the site of study on the River Stiffkey was much shorter (c. 4 km), a similar pattern was observed.

### 3.4.3 Boom or bust: location and construction of rehabilitation gravels

Although there has been an insurgence of rehabilitation projects over the last couple of decades in the UK, most of these projects have been aimed at the reach-level and as such cannot contribute meaningful ecological recovery (for target species) at the catchment level (Walker at al., 2002). It has been recognised elsewhere that installation of rehabilitation gravels as a salmonid spawning habitat into a stream characterised by high fine sediment loading is ineffective (Montgomery and Bolton, 2003).

For reach sustainability, rehabilitation gravels should not alter sediment transport dynamics. Rehabilitation gravel spacing and amplitude control water level profiles and therefore channel processes at this scale (Walker et al., 2004). Consequently accurate determination of these factors is key to sustainable management strategies that involve rehabilitation gravels. The installation of rehabilitation gravels into artificially deepened reaches of the River Stiffkey altered local channel processes, reduced upstream transport capacity and encouraged sediment disposition. Harper et al. (1998) examined rehabilitation gravels installed into artificially straightened and deepened reaches in Harpers Brook, a tributary of the River Nene that drains agricultural land in eastern England. Within 3 years >30% were dominated by sediment (<2 mm) and were considered ineffective spawning habitat. This failure was attributed to a combination of factors; the location of gravels within the stream, poor construction and inadequate inter-gravel spacing. Walker et al. (2004) encouraged the application of geomorphology to suitably locate rehabilitation gravels within the stream and to base construction on attributes of natural spawning gravels of the receptor stream. Harper et al. (1998) observed that rehabilitation gravels geomorphically similar to natural gravels were more successful than those that were not. Location and construction of rehabilitation gravels in the River Stiffkey were not based on natural gravel habitat characteristics.

Although much of the gravel habitat has been removed by channelisation, many gravel habitat-forming processes on the River Stiffkey have been artificially constrained by morphological alteration. The occurrence of natural gravel deposits is determined by hydraulic control processes in reaches of greater gradient, high width-depth ratios and increased sinuosity (Leopold et al., 1964). However, the absence of key morphological features from planform to the macrohabitat scale modify the hydrological processes necessary for natural gravel turnover in the River Stiffkey.

### 3.5 Conclusion

Precipitation, land-use, abstraction and river modification are key catchment controls that regulate the physical character of the river, and consequently rehabilitation gravels. The River Stiffkey requires sediment control mechanisms at the catchment scale to facilitate natural recovery (Sear, 1994). Without adequate retention mechanisms, modern large-scale agricultural practice ensures a significant and readily available supply of sediment to the river channel. Rehabilitation gravels installed into the River Stiffkey were based on a species-specific habitat scheme delivered at the reach-scale (see Clarke et al., 2003). The locating of rehabilitation gravels was considered in isolation (deficiency of gravel habitat) from controlling processes. Formative sediment dynamics (sediment supply, transport and storage processes) that shape channel morphology, and acknowledgment of sediment storage time scales, appear to have been overlooked (see Sear, 1994). Rehabilitation gravels can increase habitat heterogeneity where natural riffle-pool morphology is absent. Management strategies should however be sensitive to underlying controlling processes and address environmental constraints that limit ecological recovery. The catchment-scale approach to river rehabilitation has been advocated since at least the late 20th century to recognise and account for processes operating at this scale. Boon (1998) and Harper et al. (1999) both advocate a holistic catchment scale approach that facilitates variable spatial and temporal scales of recovery. Both Sear (1994) and Montgomery and Bolton (2003) stress the importance of geomorphology as the mechanism to identify catchment specific processes and constraints to ecological recovery that underpin the rehabilitation framework. Constraints identified at the catchment scale are key determinants of river ecosystem processes. Management strategies should consider these constraints to rehabilitation, addressing large-scale controlling factors prior to rehabilitating habitat diversity at the reach scale. Reinstating vital catchment processes in key channel reaches either in combination with rehabilitation gravels or without is a fundamental parameter in the design of management approaches in streams with high sediment load pressures. Clarke et al. (2003) argued that a multidisciplinary and process-led framework delivered at the catchment-scale underpins sustainable rehabilitation.

# 4 The microhabitat character of rehabilitation gravels: a sedimentological analysis

## 4.1 Introduction

The sedimentological structure and composition of spawning gravels, whether natural or artificially introduced, are important factors that directly influence *S. trutta* population recruitment at the embryo life-stage. Determination of sediment grain-size distribution and composition provides a measure of spawning habitat suitability (Lotspeich and Everest, 1981; Acornley and Sear, 1999; Kondolf, 2000). Inadequate sediment within the range identified as suitable for spawning (64>$D$≥16 mm) or an abundance of fine sediment (<1 mm) can reduce spawning and embryo development appreciably (Kondolf and Wolman, 1993; Armstrong et al., 2003; Greig et al., 2005a). This chapter examines the physical characteristics, specifically the morphosedimentary nature, and the associated suitability of rehabilitation gravels as a *S. trutta* spawning habitat. This chapter has three key objectives:

- to examine the sediment structure and composition of rehabilitation gravels in view of the wider physical context
- to examine the physical suitability of rehabilitation gravels for migratory and non-migratory *S. trutta* spawning
- to determine whether rehabilitation gravels undergo a physical morphological succession over the short to medium-term. This is based on significant changes in grain-size and fine sediment concentration between gravel sites installed to the same specifications in 2003 and in 2009.

This chapter reports the morphosedimentary impacts of those catchment controls identified in Chapter 3. Catchment controls determine the problems most commonly associated with rehabilitation gravels; erosion of spawning gravels during high flows and deposition of fine grained sediments into gravel interstices. In order to investigate the sediment composition of rehabilitation gravels in the River Stiffkey, freeze cores, a gravel survey, stream velocity and non-migratory *S. trutta* survey data were examined.

Four 2009 rehabilitation gravels (2009A, D, F and J), all three 2003 rehabilitation gravels (2003A-C) and three naturally occurring gravel sites identified as spawning habitat and used as controls (Whey Curd, Water Hall and Fort) (see Figure 2.1 Chapter 2) were sampled by freeze coring. Freeze coring kept the fine sediment fraction of the spawning substrate in-situ and provided a vertical grain-size distribution profile that was examined at 5 cm depth intervals (section 2.3.2, Chapter 2). Freeze cores were sampled at the upstream and downstream extent as well as the mid-point of the gravel site. This non-random sampling design ensured greater coverage at each site. Freeze cores were sampled at a minimum depth of 30 cm. This ensured that spawning sediments representative of shallow spawning non-migratory *S. trutta* of Norfolk (Milan et al., 2000) as well as migratory *S. trutta* (30 cm) (Crisp and Carling, 1989) were included. For the purposes of *S. trutta* spawning, the sediment composition was examined for the following grain-size ranges: gravel (64>$D$≥16 mm), coarse sand (2>$D$≥1 mm), clay particles (<0.004 mm), finer grained sediment (<1 mm) and median grain-size ($D_{50}$). Cobbles ≥64 mm were described based on axis dimensions (mm), with roundness based on the index of Powers (1953) and percentage weight contribution to grain-size distribution. A threshold value of 14% fine sediment (<1 mm) was used as a proxy of a healthy spawning gravel deposit that enabled 50% alevin emergence, based on Milan et al. (2000). The sand index (SI), a measure of the composition of sand in spawning substrate, was determined as a further quality index. A SI value smaller than 1 has been shown to be excellent for *S. trutta* emergence, whilst 1.5 is indicative of a poor habitat restrictive for alevin emergence (Peterson and Metcalfe, 1981).

A quantitative survey of stream-bed gravel (≥5 mm) was conducted to determine naturally available spawning grain-sizes, and to investigate how the introduction of rehabilitation gravels impacted spawning gravel abundance for migratory and non-migratory *S. trutta*. Transects across the stream width were taken at 7 m intervals throughout the study site. Three surface gravel samples were taken at 0.25%, 0.5% and 0.75% channel width using a garden trowel and graded in the field using a customised gravelometer (Figure 2.4, Chapter 2). This was a 200 mm$^2$ polyvinyl chloride (PVC) board used to grade gravel at 5, 10, 15, 20, 30, 40, 50, 60, 70, 80, 90 and 100 mm. Gravel 70≥$D$≥10 mm and 30≥$D_{50}$≥10 mm were used as proxies for migratory *S. trutta* and non-migratory *S. trutta* respectively.

The non-migratory *S. trutta* gravel size range was based on weighted average length (10%) of sexually mature non-migratory *S. trutta* within the river (Witzel and MacCrimmon, 1983; Crisp and Carling, 1989; Kondolf and Wolman, 1993; Moir and Pasternack, 2010). This data set was derived from a collation of electric fishing survey data collected by the Environment Agency (EA) and the Hull International Fisheries Institute (HIFI), University of Hull (see section 2.3.2.5, Chapter 2). The mean age at which fish became sexually mature is marked by a sharp decline in annual growth rates as greater investment is placed in gonad development than it is in somatic growth (Ricker, 1975). Water velocity was monitored during July-October of 2010, 2011 and 2012 for all gravel sites. Velocity was measured at 60% depth and 5 cm above the stream-bed using a Valeport Braystoke BFM002 miniature current flow meter. Mean water velocity provided an indication of near-bed conditions that influenced sediment transport and deposition at spawning sites.

### 4.2 Overview of rehabilitation gravel construction

Rehabilitation spawning gravels were constructed by overlaying an anchoring layer of cobbles and small boulders (100-174 mm) with flint rejects and coarse gravels (10-40 mm) (T. Jacklin, pers. comm., 17/01/2011). The artificial bedforms span bank to bank, approximately 5 m, ranging between 15-40 m in length (Figure 4.1) and formed two crude pool-riffle sequences consisting of three and four riffles each, as well as several other isolated gravel bedforms (see Figure 2.1, Chapter 2). Three rehabilitation gravels were installed in 2003 and a further 10 in 2009. The uniform channel geometry was altered post-gravel introduction. Rehabilitation gravels were installed in straightened, over deepened channels creating a series of distinct ingot-like bedform features that modified the longitudinal channel planform (Figure 3.17, Chapter 3). Rehabilitation gravels have steep unstable downstream ends and fine sediment supported upstream ends. The deep channel, once dominated by sand, now has greater physical heterogeneity (albeit over the short term), a more spatially variable velocity and a generally coarser substrate. A long reach of unmodified stream-bed splits the rehabilitation gravel instalments into two distinct groups (see section 6.4, Chapter 6). A suitable natural spawning gravel site, Fort, within the unmodified stream-bed reach, and a further two natural gravel sites upstream of rehabilitation gravels, where characterised by homogonous shallow stream depths. Unlike rehabilitation gravels, natural spawning gravels were small and spatially fragmented patches within a continuous gravel-framework matrix-filled streambed (Figure 4.2). Natural pool-riffle sequences are poorly defined throughout the study site.

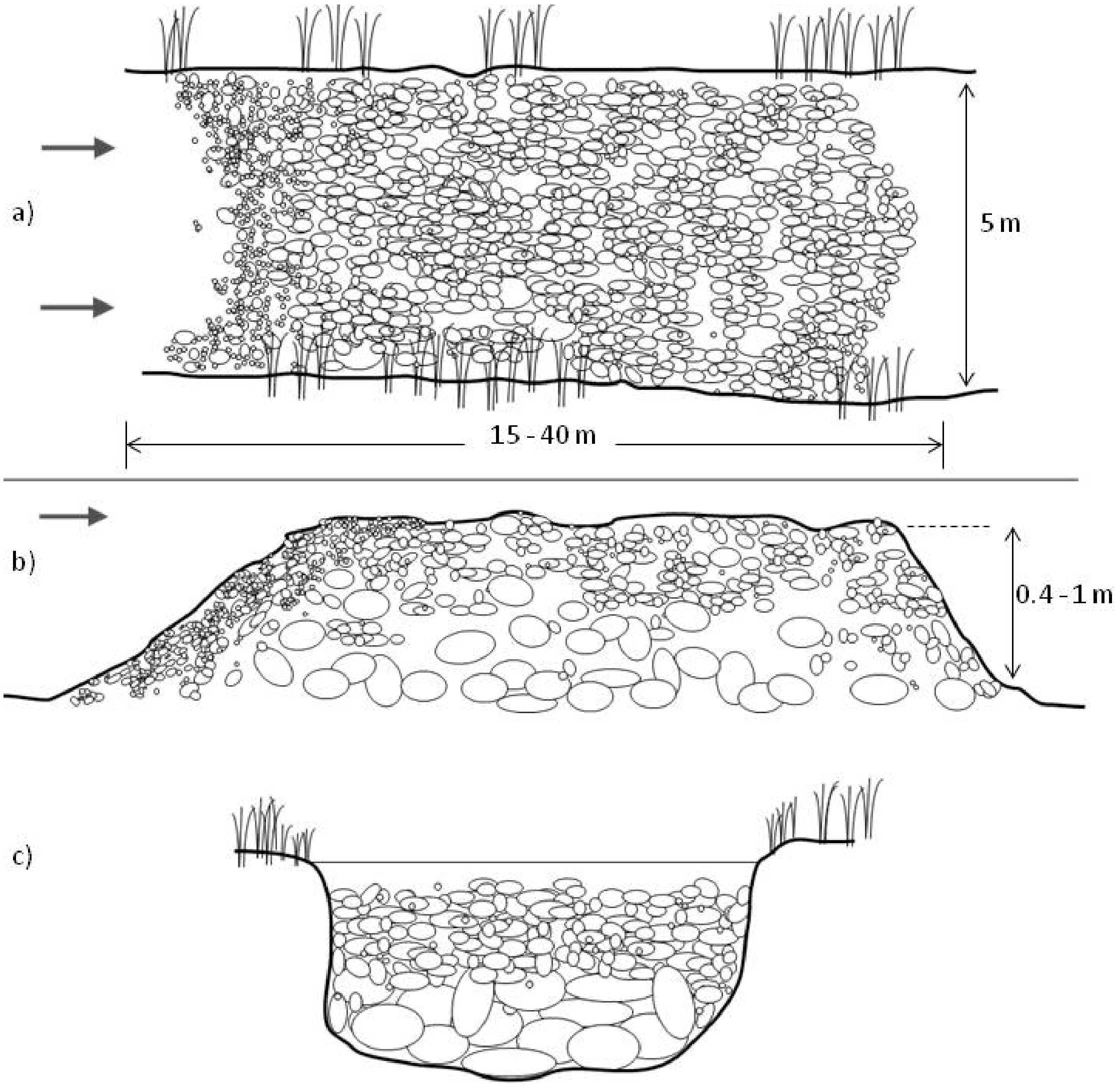



**Figure 4.1** Plan view (a), side view (b) and section view (c) schematic showing the structure of rehabilitation gravel within the River Stiffkey channel. Rehabilitation gravel was installed as flat bank-to-bank structures filling in short sections of the over-deepened channel reaches.

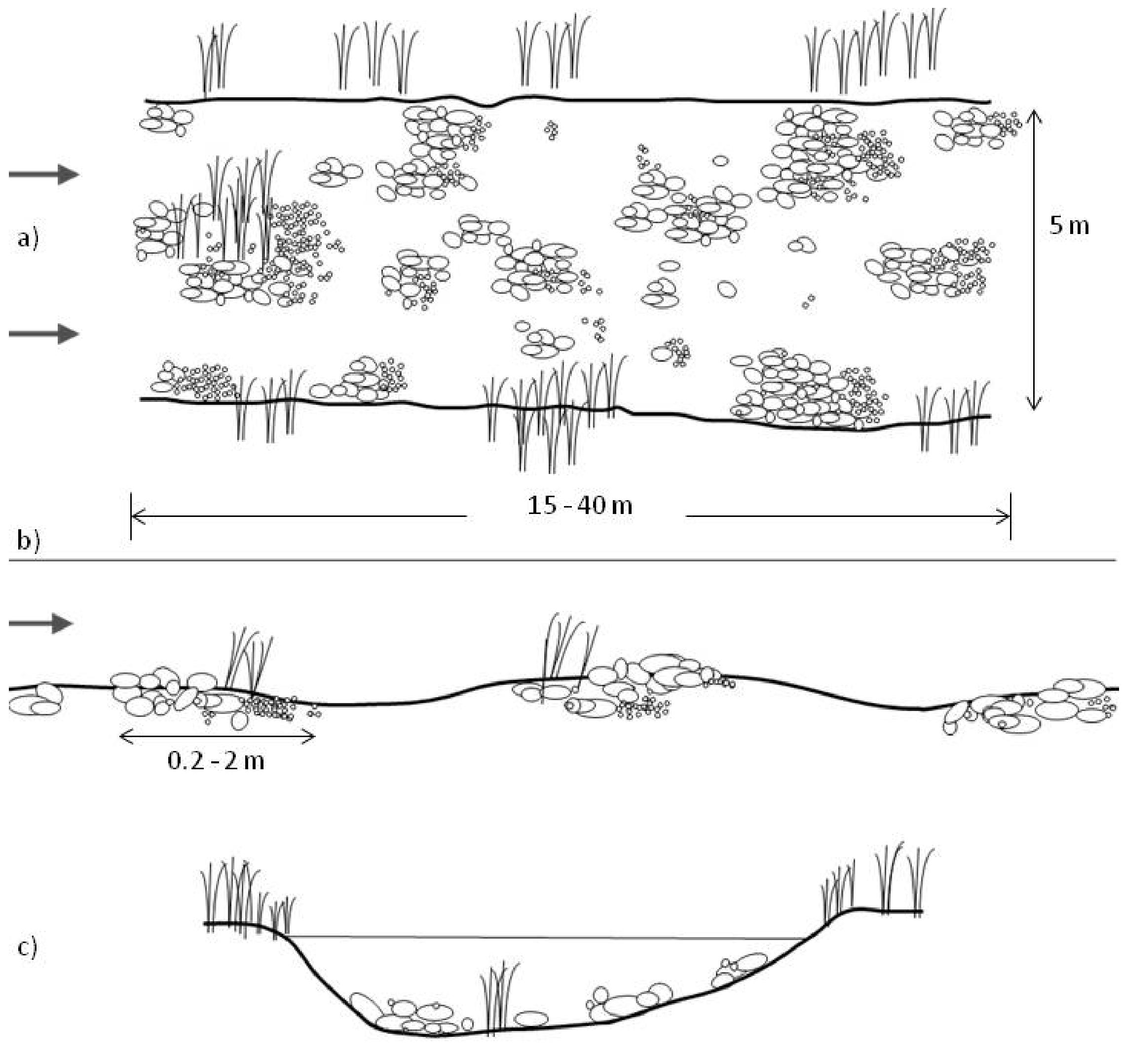



**Figure 4.2** Plan view (a), side view (b) and section view (c) of an idealised schematic of naturally occurring gravels within the River Stiffkey channel. Natural gravels exhibit poor sorting and were spatially fragmented.

### 4.3 Rehabilitation gravel characteristics: analysis of *S. trutta* spawning sediments

A total of 47.1 kg of sediment was examined from all cores sampled from the 2003 rehabilitation gravel treatment, 57.5 kg from the 2009 rehabilitation gravel treatment and 31.2 kg of natural gravel treatment, with a total combined weight of 135.8 kg (Table 4.1). There was greater uncertainty, based on Church et al. (1987), if freeze core $D_{50}$ approximates population $D_{50}$ for individual freeze core samples as well as combined freeze cores at site and treatment level, than the less stringent criteria of Mosely and Tindale (1985), who recommended a total weight 6 kg. However, based on 5% composition of 64 mm particle size (Mosely and Tindale, 1985), sample error and uncertainty decreased as individual freeze core weights of rehabilitation and natural gravel were combined at the site and treatment scale to well over 6 kg (Table 4.1). Uncertainty of poorly sorted natural gravel however was great based on sample weight criteria reported by Milan et al (1999). As outlined in Chapter 2, such error was implicit. The 2003 and 2009 treatments had similar cumulative percentage grain-size weights, whilst the natural treatment was compositionally distinct (Figure 4.3; Mann-Whitney, $p<0.05$, Table 4.2). Moreover, sites within the natural treatment had significantly distinct percentage grain-size weights (Mann-Whitney, $p<0.05$, Table 4.2). Fort and Whey Curd had the lowest gravel composition and greater percentages of silt ($D<0.0063$ mm), whilst Water Hall contained good percentages of gravel ($64>D\geq2$ mm) but also sand ($2>D\geq0.063$ mm). In contrast, sites within the 2003 rehabilitation gravels had comparable sediment compositions. The 2009 treatment displayed sediment composition variability between rehabilitation gravel*s* (Kruskal-Wallis, $p<0.05$, Table 4.2), however not all were compositionally distinct; sites 2009A and 2009J as well as 2009D and 2009F had similar sediment distributions. The natural treatment contained a significantly low percentage of gravel within the range $64>D\geq16$ mm, whilst both rehabilitation treatments contained significantly greater gravel contributions (Mann-Whitney, $p<0.05$, Table 4.3). However, gravel $64>D\geq16$ mm was comparable between sites within each treatment.

Establishing an ideal *S. trutta* spawning gravel size range is important for rehabilitation gravel projects to achieve the greatest population recruitment potential. *S. trutta* length is positively correlated with size of spawning gravel; 10% of fish length determines a suitable grain-size $D_{50}$ (Kondolf and Wolman, 1993). Therefore an ideal spawning gravel size range was based on determining 10% of the weighted average length of all sexually mature non-migratory *S. trutta* within the River Stiffkey. In order to achieve this, growth rates of the resident non-migratory *S. trutta* population were determined. The mean age at which fish became sexually mature is marked by the sharp decline in annual growth rates as greater investment is placed in gonad development than it is in somatic growth (Ricker, 1975). *S. trutta* growth rates were based on a collation of surveys conducted periodically between 2000-2011 by the EA and HIFI (see section 2.3.2.5, Chapter 2). An increase in annual growth rates ($G_r$) from the 0+ to 1+ year class was observed whilst decreases in $G_r$ between 1+ and 2+, as well as 2+ and 3+ fish, indicated greater investment in gonad development and less investment in somatic growth (Ricker, 1975; Hendry and Berg, 1999) (Figure 4.4). Therefore the earliest onset of sexual maturity, indicated by the initial reduction in $G_r$, was evident in the 2+ year class fish. Weighted average length of all sexually mature fish age 2+ and older was 200.6 mm. The naturally required spawning gravel $D_{50}$ for non-migratory *S. trutta* was therefore 20.06 mm (10% of body length), and associated with sediment retained on sieves between the $30>D\geq16$ mm size range. This sediment size range ($30>D_{50}\geq16$ mm) was used in combination with $64>D\geq16$ mm, $2>D\geq1$ mm, $D<1$ mm, $D<0.004$ mm to describe *S. trutta* spawning suitability of rehabilitation gravels.

**Table 4.1** Summary table of individual freeze-core, site (cumulative core) and treatment (cumulative site) sample weights.

| Treatment | Site | Core | Core (kg) | Site (kg) | Treatment (kg) |
|---|---|---|---|---|---|
| 2003 | 2003A | 1 | 3.31 | | |
| | | 2 | 4.93 | | |
| | | 3 | 2.01 | 10.25 | |
| | 2003B | 1 | 8.42 | | |
| | | 2 | 2.25 | | |
| | | 3 | 5.37 | 16.04 | |
| | 2003C | 1 | 10.47 | | |
| | | 2 | 7.16 | | |
| | | 3 | 3.16 | 20.79 | 47.1 |
| 2009 | 2009A | 1 | 1.85 | | |
| | | 2 | 12.79 | | |
| | | 3 | 2.59 | 17.23 | |
| | 2009D | 1 | 6.53 | | |
| | | 2 | 3.01 | | |
| | | 3 | 2.47 | 12.01 | |
| | 2009F | 1 | 2.10 | | |
| | | 2 | 3.32 | | |
| | | 3 | 2.37 | 7.79 | |
| | 2009J | 1 | 3.16 | | |
| | | 2 | 6.29 | | |
| | | 3 | 11 | 20.43 | 57.5 |
| Natural | Fort | 1 | 3.31 | | |
| | | 2 | 4.93 | | |
| | | 3 | 2.01 | 10.25 | |
| | Water Hall | 1 | 3.30 | | |
| | | 2 | 5.28 | | |
| | | 3 | 3.44 | 12.02 | |
| | Whey Curd | 1 | 3.32 | | |
| | | 2 | 2.8 | | |
| | | 3 | 2.74 | 8.88 | 31.2 |

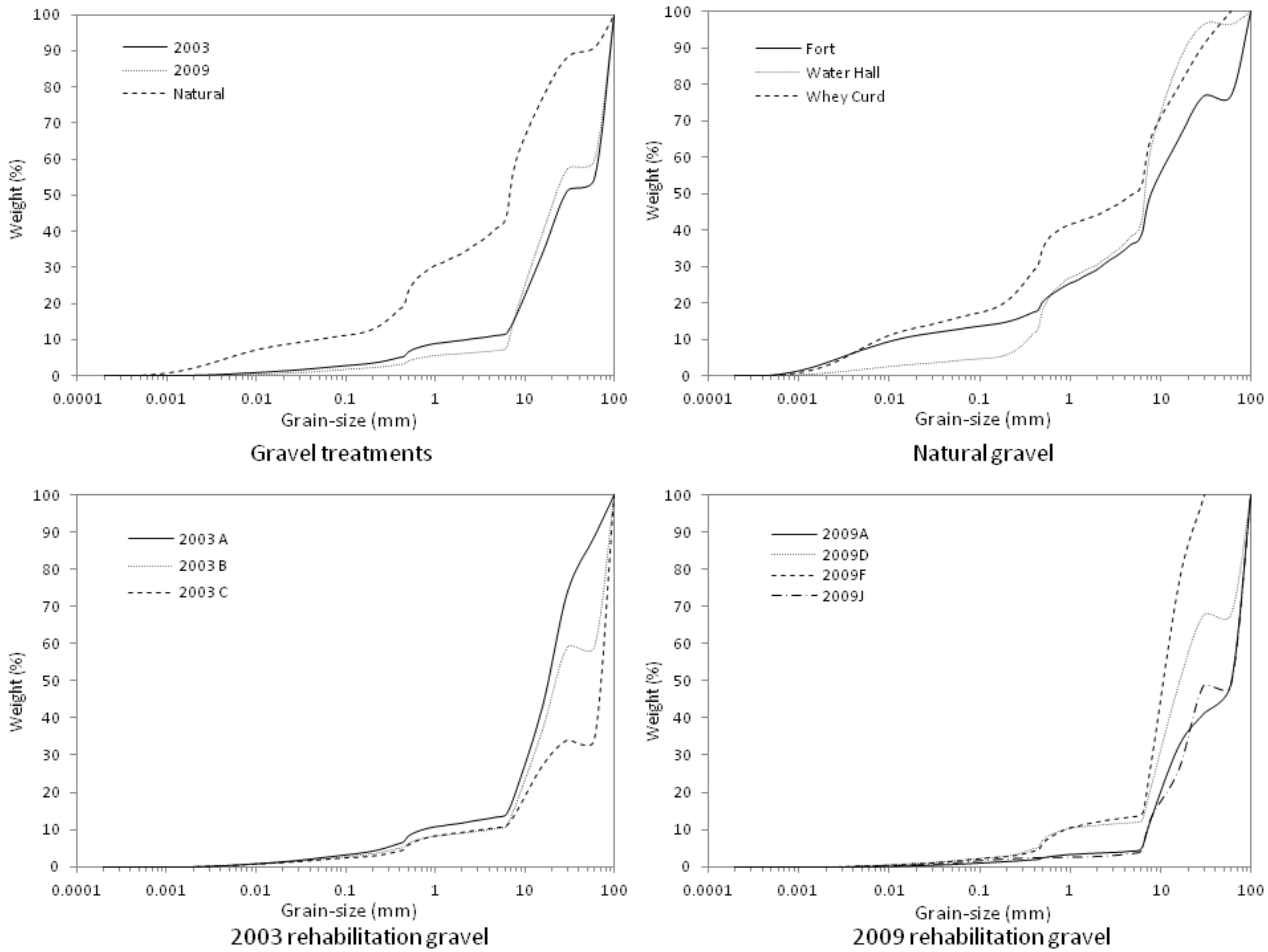



**Figure 4.3** Cumulative grain-size curves for gravel treatments. Rehabilitaion gravel treatments were less well sorted with a greater composition of coarse material. Natural gravel illustrated greater variability with a higher composition of matrix material than observed in rehabilitation gravel.

Sediments suitable for non-migratory *S. trutta* spawning ($30>D_{50}\geq16$ mm) were found at comparable percentages between all three gravel treatments. Greater variation was observed between sites within the natural and 2009 treatments (Kruskal-Wallis, $p<0.05$, Table 4.3). Substrate at the Water Hall natural site contained a greater percentage of non-migratory spawning gravel ($30>D_{50}\geq16$ mm) than the other natural treatment sites, whilst the Whey Curd site recorded the least (Kruskal-Wallis, $p<0.05$, Table 4.3). The percentage of gravel $30>D_{50}\geq16$ mm was comparable between the 2003 rehabilitation gravel sites. Overall composition of fine sediment (<1 mm) varied significantly between treatments (Kruskal-Wallis, $p<0.05$, Table 4.3). A significantly greater percentage of sediment $D<1$ mm was observed in the natural treatment compared to the 2009 treatment (Figure 4.3; Mann-Whitney, $p<0.05$, Table 4.3). Furthermore, greater percentages of finer grained sediments (<1 mm) occurred in the 2003 gravel treatment than the 2009 gravel treatment, but less than within natural treatment. Greater percentages of sediment $D<1$ mm, however, were associated with increased depth across all treatments (Figure 4.5). Both 2003 rehabilitation gravels and naturally occurring substrate frequently exceeded the 14% threshold of sediments $D<1$ mm that inhibit alevin emergence.

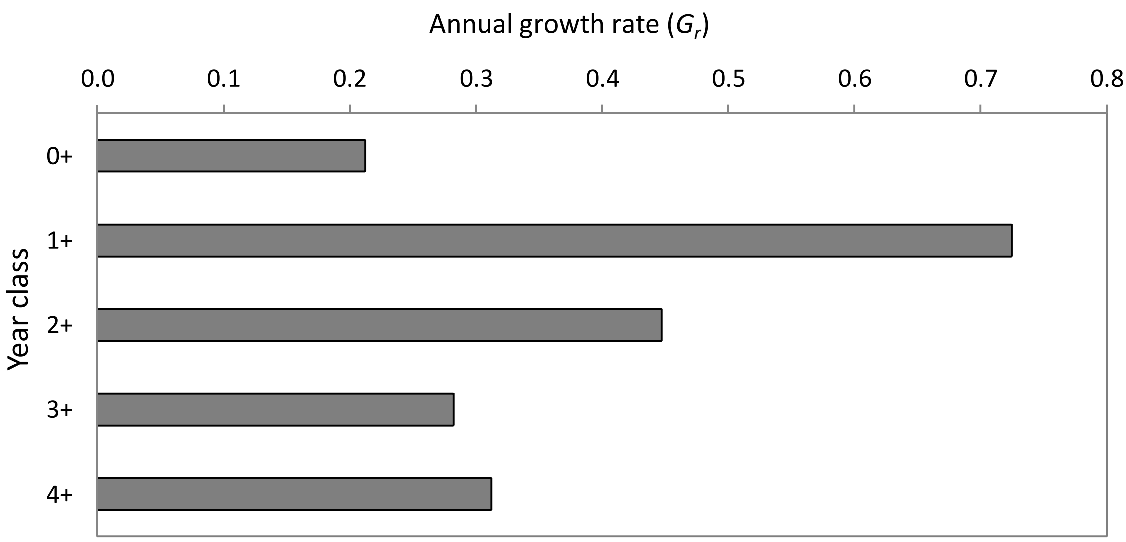



**Figure 4.4** Annual growth rate (*Gr*) of non-migratory *S. trutta* derived from mean year class fork-length. Bars represent *Gr* for each year class. The earliest decline in annual growth, at 2+, indicated a greater investment in gonad development and therefore the onset of sexual maturity (Hendry and Berg, 1999).

**Table 4.2** Summary results of the Kruskal-Wallis and Mann-Whitney U analysis for percentage grain-size composition difference between gravel treatments, sites (s) and cores (c). *1* indicates a significant or positive test result, *0* indicates a negative result and - indicates no test.

| | Kruskal-Wallis | Mann-Whitney U | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | 2003 | 2009 | Water Hall | Whey Curd | 2003B | 2003C | 2009D | 2009F | 2009J |
| **Treatment** | 1 | - | - | - | - | - | - | - | - | - |
| **Natural (s)** | 1 | 1 | 1 | - | - | - | - | - | - | - |
| **Fort (c)** | 1 | - | - | 1 | 1 | - | - | - | - | - |
| **Water Hall (c)** | 1 | - | - | - | 1 | - | - | - | - | - |
| **Whey Curd (c)** | 1 | - | - | - | - | - | - | - | - | - |
| **2003 (s)** | 0 | - | 0 | - | - | - | - | - | - | - |
| **2003A (c)** | 0 | - | - | - | - | - | - | - | - | - |
| **2003B (c)** | 1 | - | - | - | - | - | - | - | - | - |
| **2003C (c)** | 0 | - | - | - | - | - | - | - | - | - |
| **2009 (s)** | 1 | - | - | - | - | - | - | - | - | - |
| **2009A (c)** | 1 | - | - | - | - | - | - | 1 | 1 | 0 |
| **2009D (c)** | 1 | - | - | - | - | - | - | - | 0 | 1 |
| **2009F (c)** | 0 | - | - | - | - | - | - | - | - | 1 |
| **2009J (c)** | 1 | - | - | - | - | - | - | - | - | - |

**Table 4.3** Summary results of statistical analysis for percentage composition difference of gravel (64>$D$≥16 mm and 30>$D_{50}$≥16 mm) and fine sediment ($D$<1 mm) between gravel treatments, sites (s) and cores (c). 1 indicates a significant or positive test result, 0 indicates a negative result and - indicates no test. No Mann-Whitney U tests were conducted if the Kruskal-Wallis analysis had a negative result.

| | **64>$D$≥16 mm** | | | **30>$D_{50}$≥16 mm** | **$D$<1 mm** | | |
|---|---|---|---|---|---|---|---|
| | **Kruskal-Wallis** | **Mann-Whitney U** | | **Kruskal-Wallis** | **Kruskal-Wallis** | **Mann-Whitney U** | |
| | | **2003** | **2009** | | | **2003** | **2009** |
| **Treatment** | 1 | - | - | 0 | 1 | - | - |
| **Natural (s)** | 0 | 1 | 1 | 1 | 0 | 0 | 1 |
| **Fort (c)** | - | - | - | 0 | 1 | - | - |
| **Water Hall (c)** | - | - | - | 0 | 0 | - | - |
| **Whey Curd (c)** | - | - | - | 0 | 0 | - | - |
| **2003 (s)** | 0 | - | 0 | 0 | 0 | - | 0 |
| **2003A (c)** | - | - | - | 1 | 0 | - | - |
| **2003B (c)** | - | - | - | 0 | 0 | - | - |
| **2003C (c)** | - | - | - | 1 | 0 | - | - |
| **2009 (s)** | 0 | - | - | 1 | 0 | - | - |
| **2009A (c)** | - | - | - | 1 | 0 | - | - |
| **2009D (c)** | - | - | - | 0 | 0 | - | - |
| **2009F (c)** | - | - | - | 0 | 0 | - | - |
| **2009J (c)** | - | - | - | 0 | 0 | - | - |

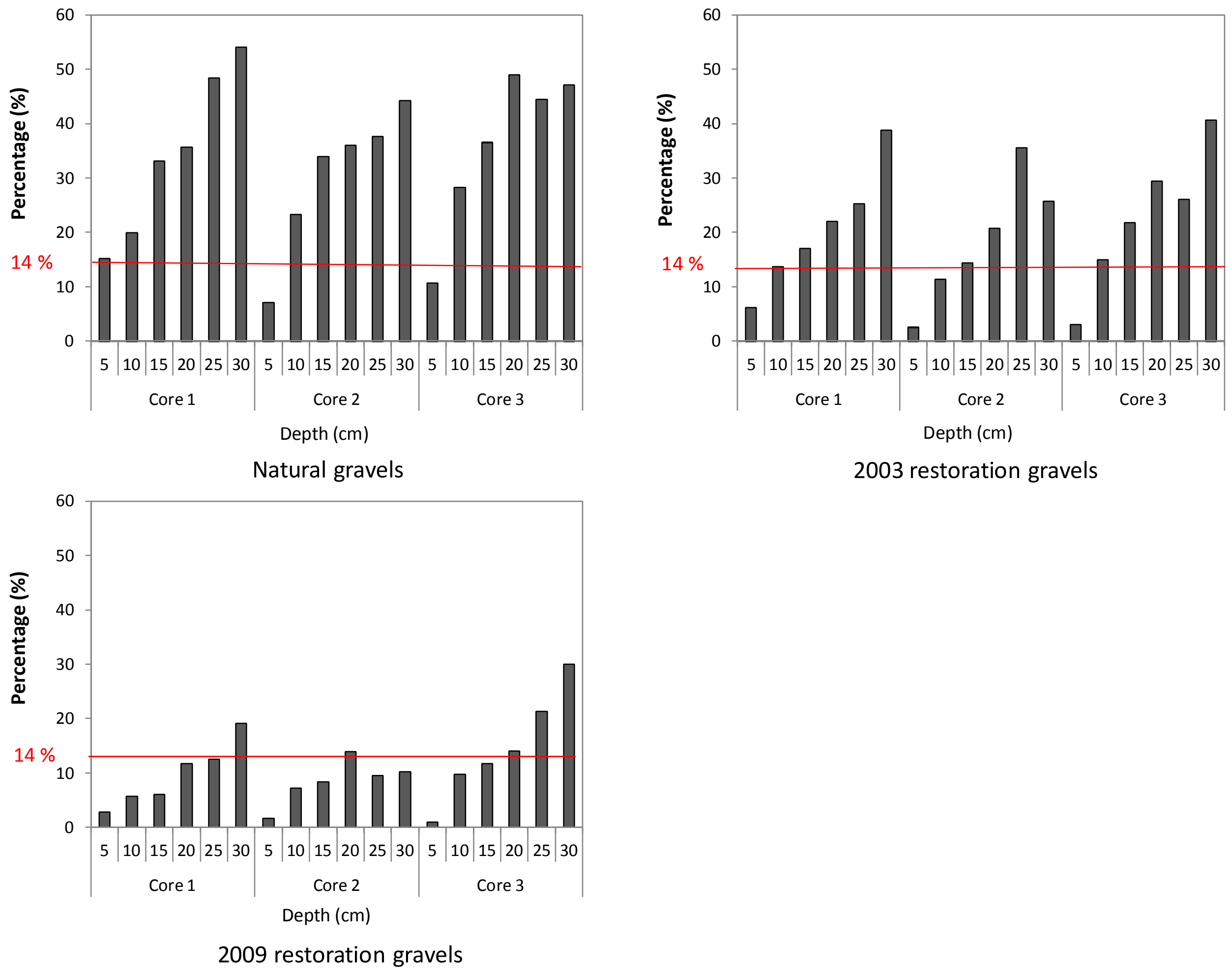



**Figure 4.5** Mean percentage sediment *D*<1 mm for each 5 cm core increment per treatment. Natural gravels exceeded the 14% threshold of sediment <1 mm for 50% embryo emergence in shallower substrate than either rehabilitation gravel treatment. The 2009 rehabilitation gravel had the least percentage sediment <1 mm in all depth increments.

### 4.3.1 Spawning sediment composition and structure at the site level

#### 4.3.1.1 Fort

The cumulative percentage grain-size weight of each core within the Fort site was significantly variable (Figure 4.6; Kruskal-Wallis, $p<0.05$, Table 4.2). The grain-size distribution was therefore spatially variable throughout the site. Surface substrate was coarser than underlying sediment creating an armour layer highly characteristic of gravel-bedded streams (Frostick et al., 1984; Reid et al., 1997; Milan et al., 2000). Spawning gravel ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) was relatively sparse below surface sediments in all cores (Figure 4.7). Substrata were characterised by a change in median grain-size diameter 10 cm below the surface as sediment $D<1$ mm became more abundant. A substantial part of the fine sediment (<1 mm) composition was clay ($D<0.004$ mm). Cobbles ($D\geq64$ mm) were observed at a depth of 15-20 cm in the upstream and midstream sections of the site, but were absent towards the downstream end of the site.

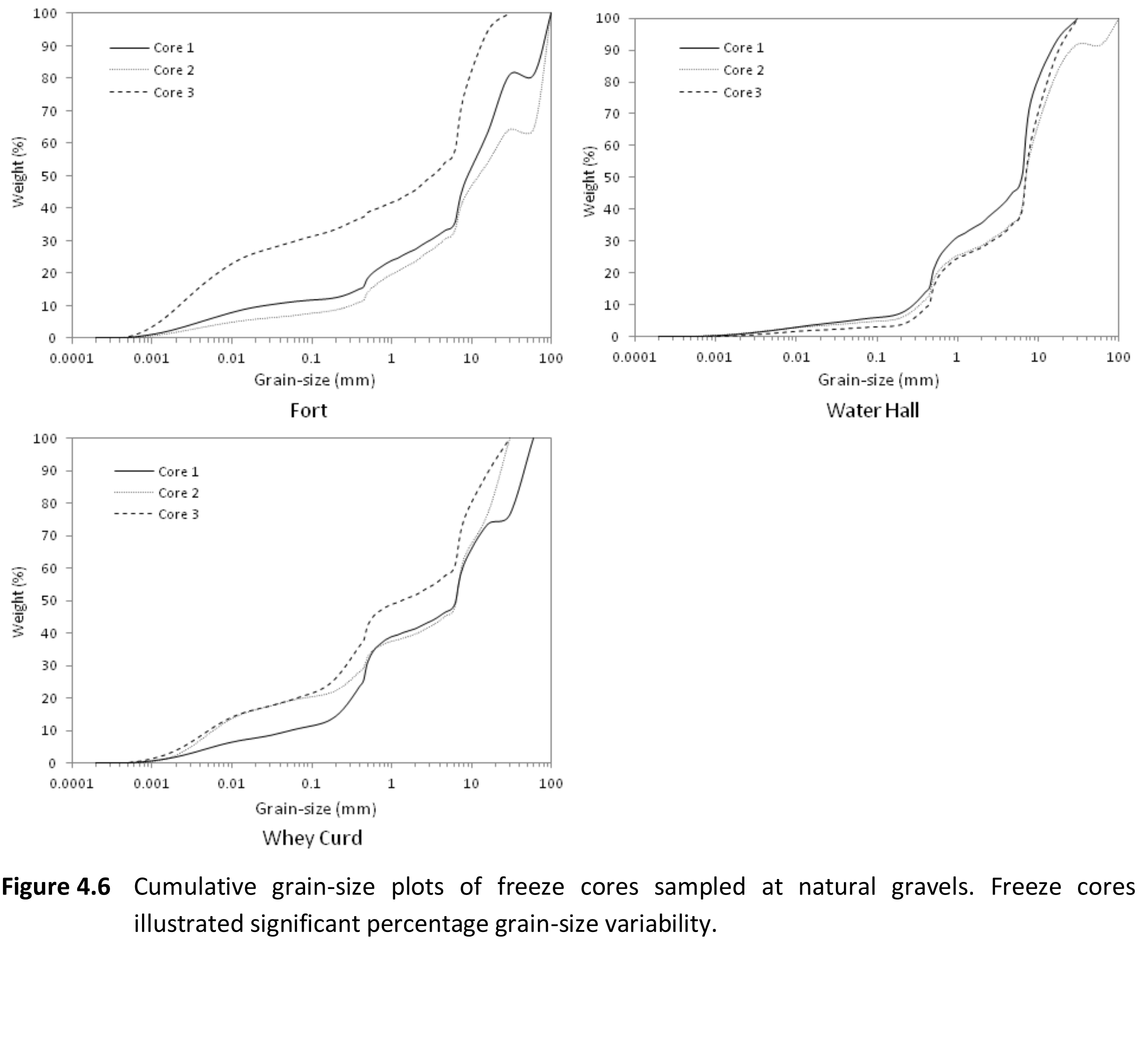



**Figure 4.6** Cumulative grain-size plots of freeze cores sampled at natural gravels. Freeze cores illustrated significant percentage grain-size variability.

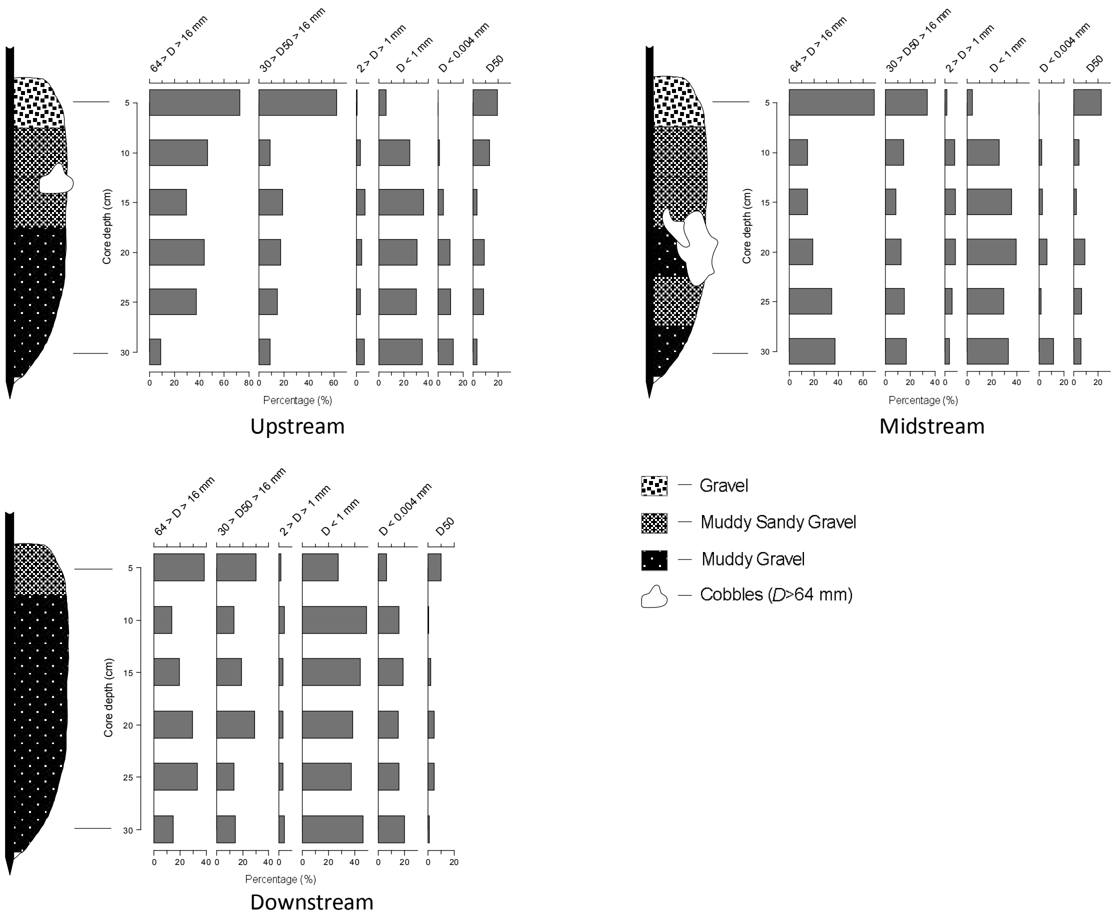



**Figure 4.7** Stratigraphy of the Fort site substrate. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D \geq 64$ mm were recovered in the upstream and midstream areas of the site.

#### 4.3.1.2 Water Hall

Cumulative percentage grain-size weight of cores were significantly variable (Figure 4.6; Kruskal-Wallis, $p<0.05$, Table 4.2), and as such sediments were spatially variable. A characteristic gravel-bed armour layer was present in surface sediments. This was apparent in the greater percentage of coarser gravels ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) in surface sediments than within the underlying substrate (Figure 4.8). As the percentage sediment (<1 mm) increased with depth, the median grain-size diameter ($D_{50}$) decreased. A single clast $D\geq64$ mm was sampled at 5-15 cm depth in the midstream section of the site.

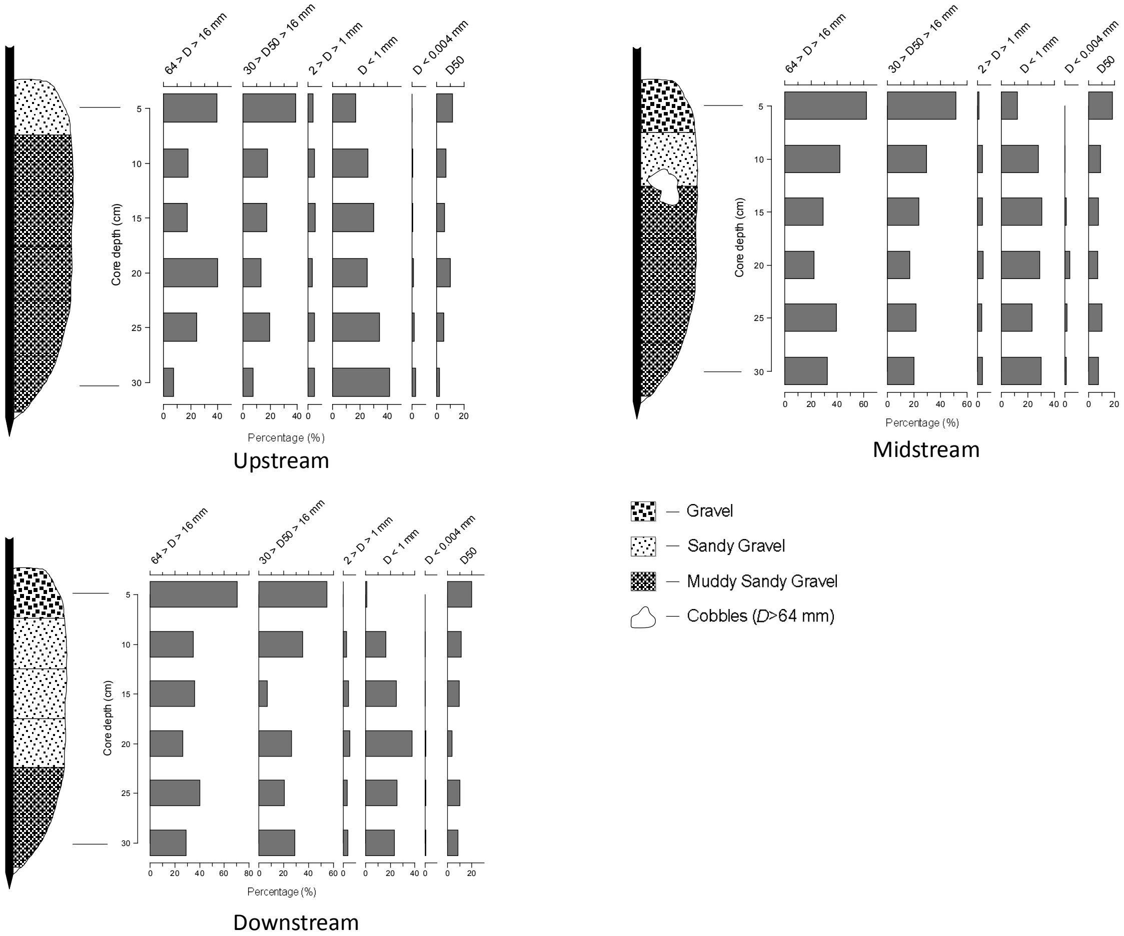



**Figure 4.8** Stratigraphy of natural substrate at the Water Hall site. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D \geq 64$ mm were recovered from the midstream area of the site only.

#### 4.3.1.3 Whey Curd

Like other sites within the natural treatment, the percentage cumulative weight of each core at the Whey Curd site was significantly different and thus indicated that the grain-size distribution was spatially variable (Figure 4.6; Kruskal-Wallis, $p<0.05$, Table 4.2). Whey Curd sediments were characterised by a surface armour layer and a substantial decrease in median grain-size ($D_{50}$) with increased depth (Figure 4.9). An abundance of fine sediment ($D<1$ mm) between 15-30 cm was a distinctive feature of the site and a cause of the substantial change in $D_{50}$. No cobbles $D \geq 64$ mm were recovered.

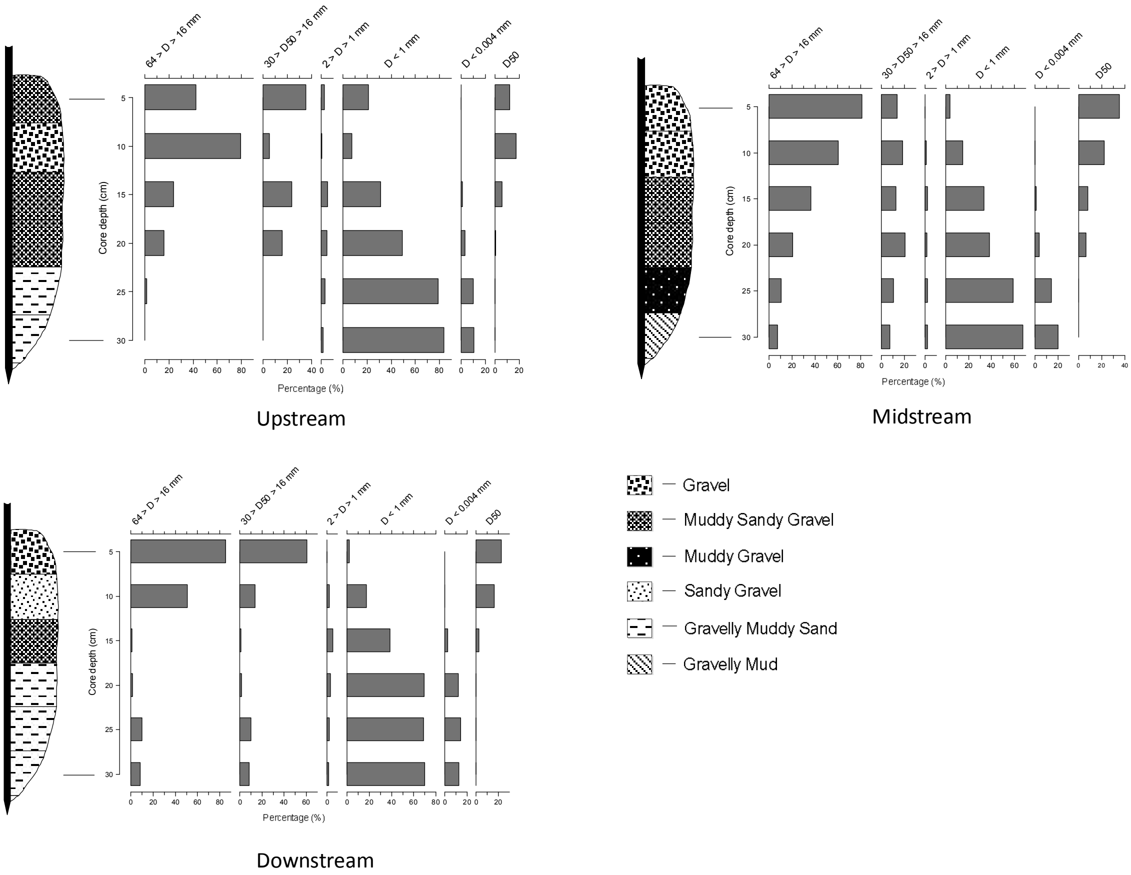



**Figure 4.9** Stratigraphy of natural substrate at the Whey Curd site. A sediment textual class was assigned to each 5 cm core depth. No cobbles $D$≥64 mm were recovered from the site.

#### 4.3.1.4 2003A

Unlike natural gravel sites, rehabilitation gravel grain-size distribution at site 2003A was relatively similar across the three cores (Figure 4.10). Sediment profiles were dominated by gravel suitable for migratory *S. trutta* (64>$D$≥16 mm) (Figure 4.11). A decrease in gravel (64>$D$≥16 mm) percentages with depth was less distinct relative to the other 2003 sites. Although there was a tendency for percentage sediment $D$<1 mm to increase in abundance within each core, the grain-size $D_{50}$ did not decrease appreciably. A surface armour layer was less apparent, particularly in the mid to upstream area of the site, than in the natural gravel sites. Erosion of the more mobile gravels in the range required by non-migratory *S. trutta* (30>$D_{50}$≥16 mm) was apparent in surface sediments across the site (Kruskal-Wallis, $p$<0.05, Table 4.3). Only a single large cobble $D$≥64 mm was sampled from surface sediments in the midstream core.

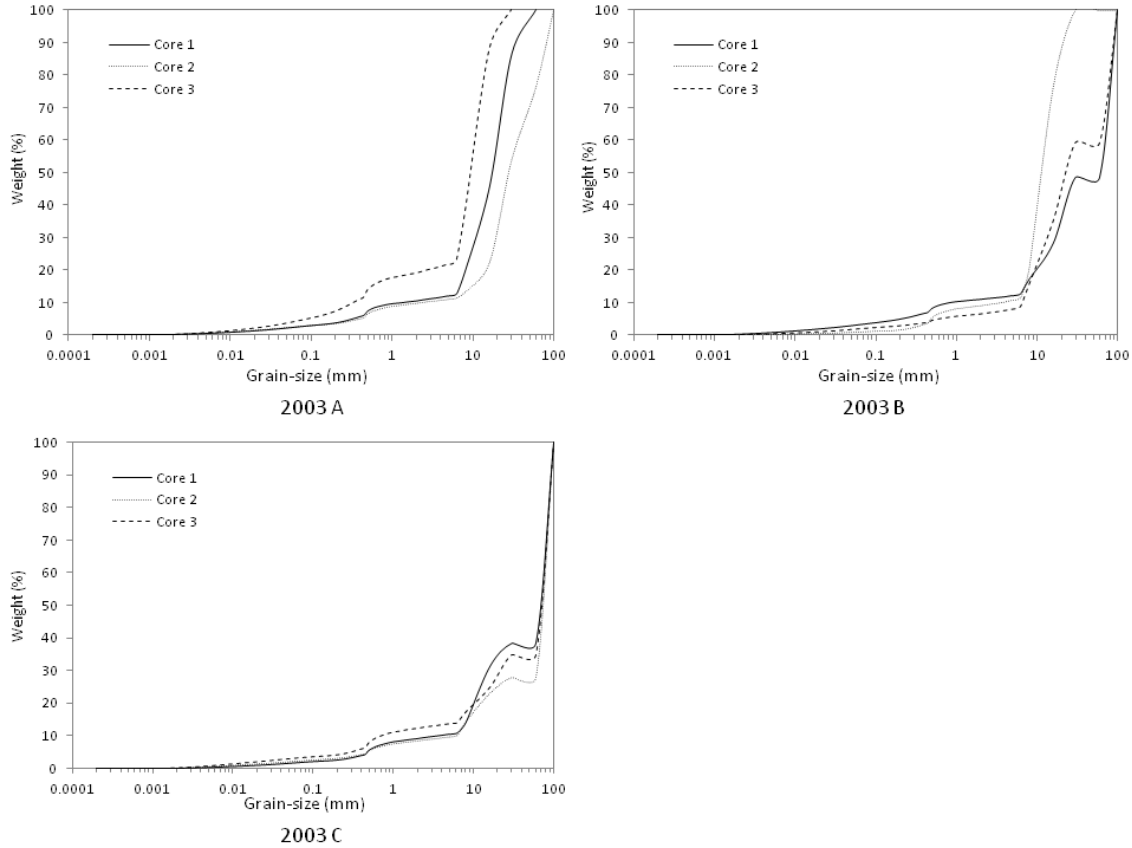



**Figure 4.10** Cumulative grain-size plots of freeze cores sampled from the 2003 rehabilitation gravel. Freeze cores taken from site 2003 B illustrated significant percentage grain-size variability.

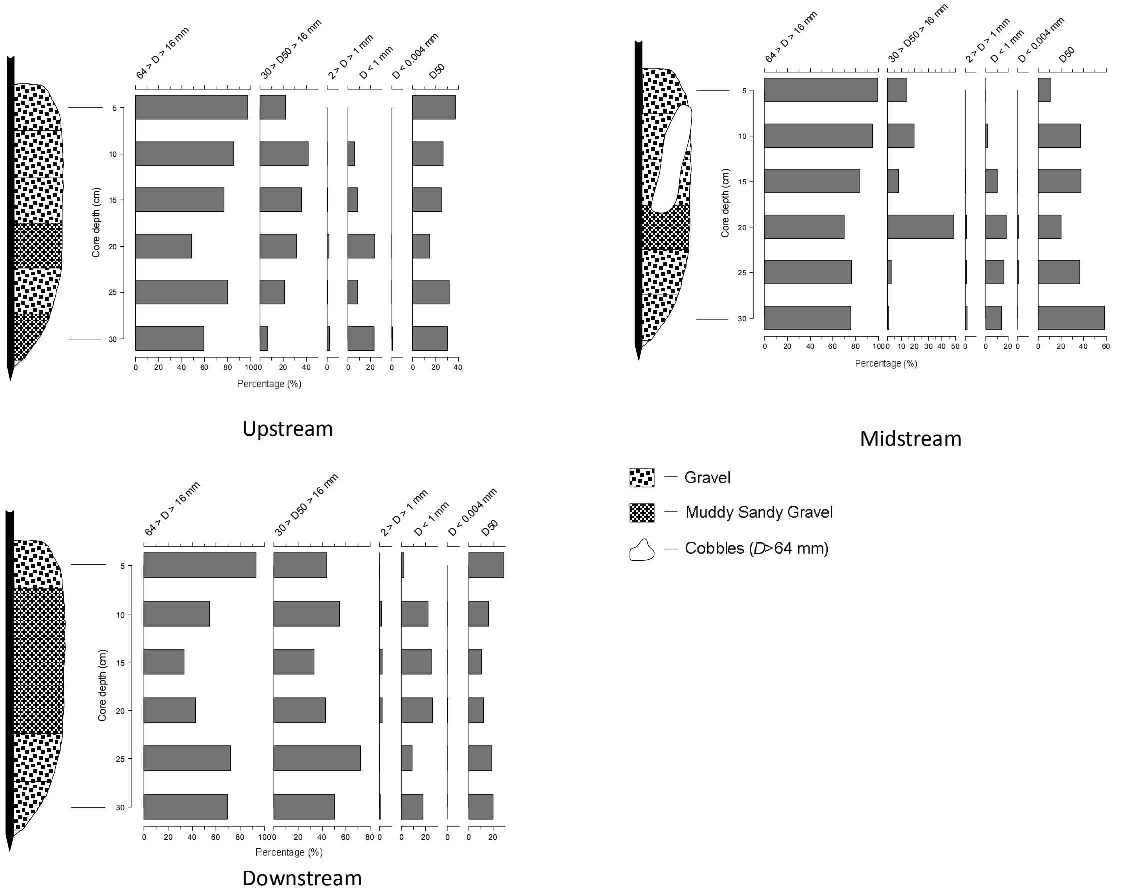



**Figure 4.11** Stratigraphy of rehabilitation substrate at the 2003A site. A sediment textual class was assigned to each 5 cm core depth. A large single cobble $D \geq 64$ mm was recovered from the midstream area of the site.

#### 4.3.1.5 2003B

The cumulative percentage grain-size weight was significantly different between each core reflecting a spatially variable grain-size distribution (Figure 4.10; Kruskal-Wallis, $p<0.05$, Table 4.2). Overall, sediment composition of the site was characterised by coarser gravel suitable for migratory *S. trutta* spawning ($64>D\geq16$ mm) and fine sediment ($D<1$ mm) (Figure 4.12). Gravel $64>D\geq16$ mm generally decreased with depth, whilst fine sediment (<1 mm) increased with depth throughout the site. The grain-size $D_{50}$ declined though the vertical extent of the upstream and downstream core but increased again at 20 cm depth for each. Sediment $D<1$ mm increased with depth in the upstream core more so then any other core. A surface armour layer was apparent in the mid and upstream core sediments. Higher percentages of gravel $30>D_{50}\geq16$ mm more suited to spawning by non-migratory *S. trutta* occurred within the mid and downstream area of the site. Erosion of surface gravel (30>$D_{50}$≥16 mm) was apparent towards the downstream end of the site. The upstream core contained a greater abundance of cobbles *D*≥64 mm in each 5 cm increment. A blockage in the tube prevented sediment adhering to the lower portion of the core extracted from the midstream area of the site.

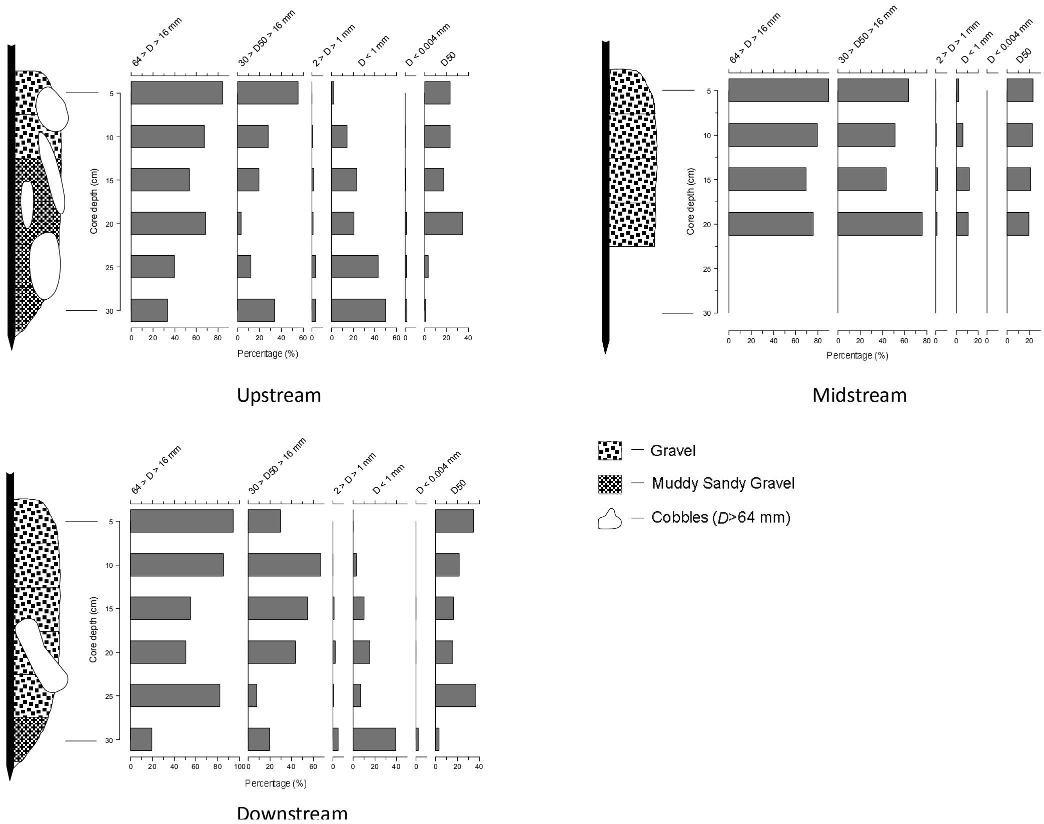



**Figure 4.12** Stratigraphy of the 2003B rehabilitation gravel site. A sediment textual class was assigned to each 5 cm core depth. Cobbles *D*≥64 mm were extracted from the upstream and downstream area of the site.

#### 4.3.1.6 2003C

Rehabilitation gravel 2003C grain-size distribution profiles were similar between the three cores (Figure 4.10). Sediments in the upstream area of the deposit were characterised by high percentages of spawning gravel (64>*D*≥16 mm and 30>$D_{50}$≥16 mm) (Figure 4.13). Percentage gravel decreased appreciably with depth throughout the mid to downstream sections of the site. The downstream core had significantly less gravel suitable for non-migratory *S. trutta* spawning (30>$D_{50}$≥16 mm) than other cores, likely due to erosion (Kruskal-Wallis, $p$<0.05, Table 4.3). Fine sediment (<1 mm) abundance increased with depth, with this especially true of the downstream end of the site. Surface armouring had developed over the spatial extent of the site. No appreciable change in median grain-size diameter ($D_{50}$) within upstream sediment was discernible. However, the $D_{50}$ decreased with increased depth in the mid and downstream section of the site. 2003C rehabilitation gravels contained relatively greater abundances of sediments within the size range 2>$D$≥1 mm than other sites within the 2003 treatment. Cobbles $D$≥64 mm were abundant throughout the mid and upstream end of the spawning habitat.

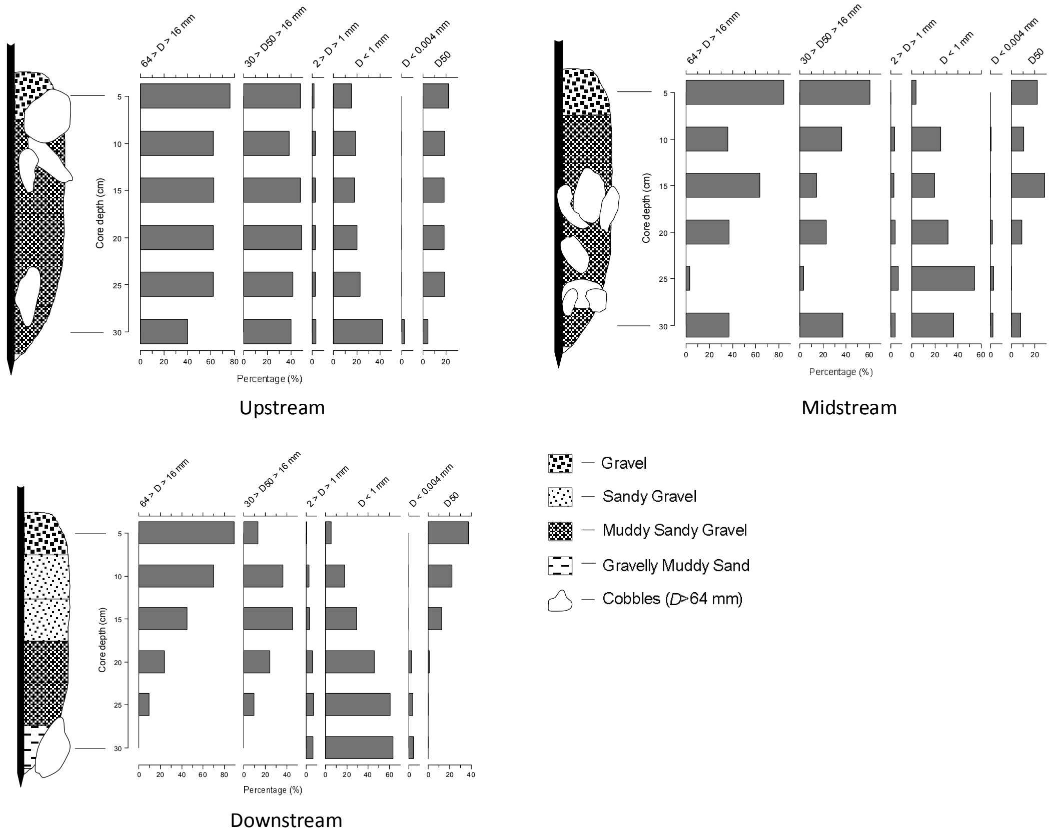



**Figure 4.13** Stratigraphy of the rehabilitation gravel site 2003C. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D$≥64 mm were extracted throughout the extent of the site.

#### 4.3.1.7 2009A

The sediment grain-size distribution was spatially variable, indicated by the significant difference between cumulative percentage grain-size weight of cores (Figure 4.14; Kruskal-Wallis, $p<0.05$, Table 4.2). Composition of spawning gravel in the ranges of $64>D\geq16$ mm and $30>D_{50}\geq16$ mm dominated this site and decreased slightly in abundance with depth (Figure 4.15). Percentage gravel remained high throughout the vertical and spatial extent of the site. Substrate of core 2 (midstream) contained a significantly lower percentage of non-migratory *S. trutta* spawning gravel $30>D_{50}\geq16$ mm than at the upstream and downstream cores (Kruskal-Wallis, $p<0.05$, Table 4.3). There was little redistribution of these more mobile gravels ($30>D_{50}\geq16$ mm) from surface substrate in the midstream area of the site. Fine sediment (<1 mm) contributed little to the composition, although percentage increased with depth in the midstream to upstream cores. Sediment $D<0.004$ mm and $2>D\geq1$ mm contributed little to sediment composition. Surface armouring was only discernible in the downstream area of the gravel deposit. Median grain-size diameter ($D_{50}$) decreased with depth at the upstream and downstream ends of the site. Cobbles $D\geq64$ mm were sampled in the lower depths from the midstream area only.

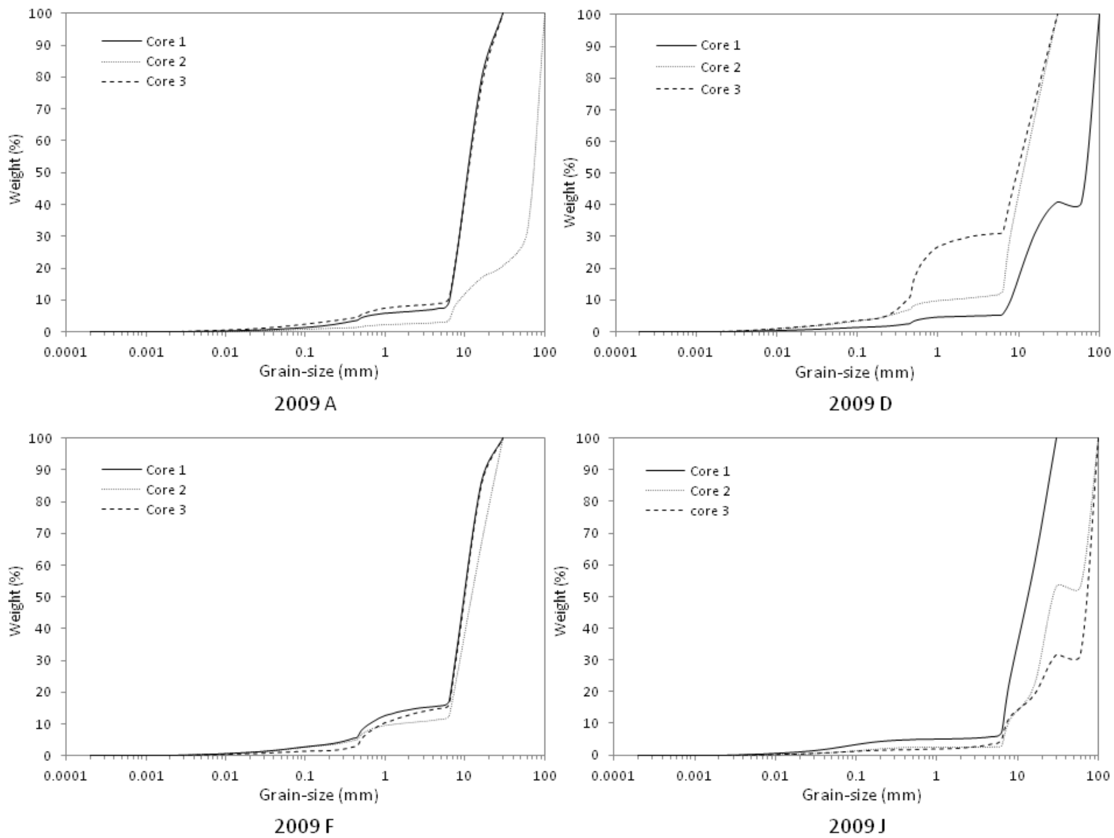



**Figure 4.14** Cumulative grain-size plots of freeze cores extracted from the 2009 rehabilitation sites. Site 2009 F had relatively less grain-size variability than any other site.

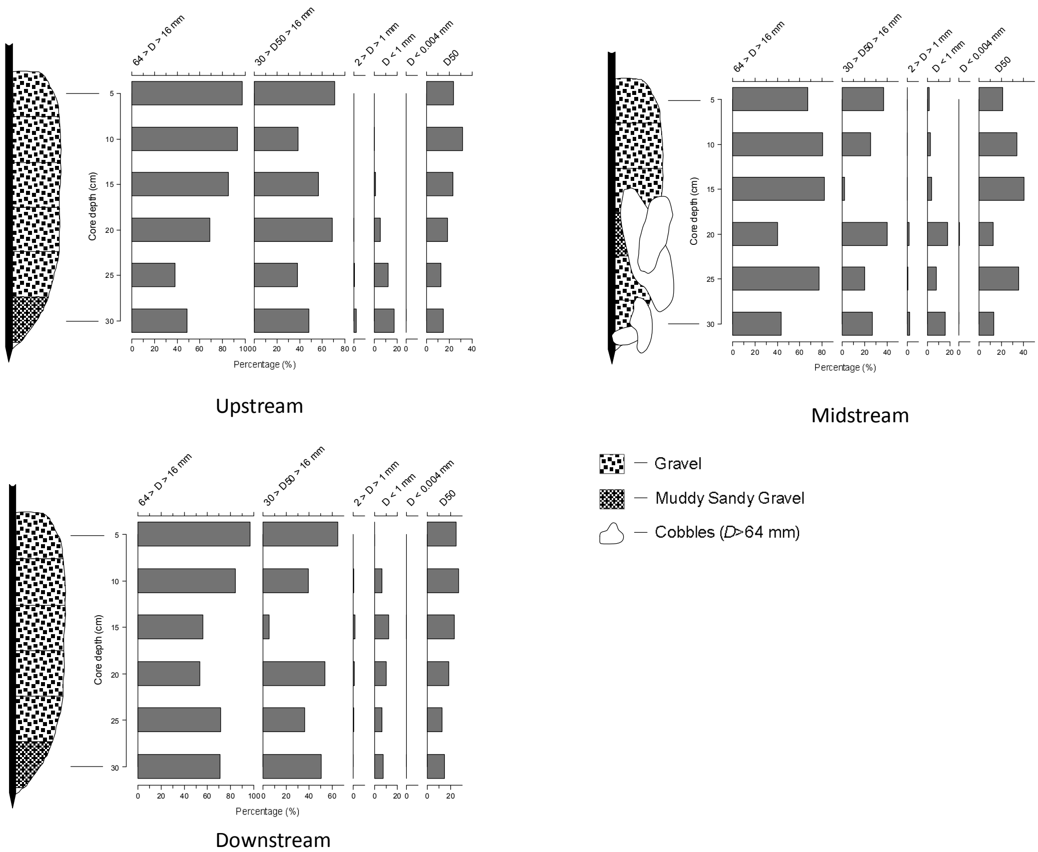



**Figure 4.15** Stratigraphy of the 2009A rehabilitation gravel site. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D \geq 64$ mm were extracted from the midstream area of the site only.

#### 4.3.1.8 2009D

2009D rehabilitation gravel grain-size distributions were spatially variable (Figure 4.14; Kruskal-Wallis, $p<0.05$, Table 4.2). There was a greater percentage of spawning gravel sediment sizes ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) in the mid and upstream sediments of the site (Figure 4.16). There was a general trend of decreasing gravel ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) percentage throughout the deposit. Surface erosion of gravels $30>D_{50}\geq16$ mm, more suited to spawning by non-migratory S*. trutta*, was evident in the upstream and midstream core. Percentages of finer grained sediment (<1 mm) increased with depth, particularly in the downstream core where there was a greater increase in percentage below 15 cm depth. Moreover, the 2009D site contained the greatest percentage of sediment $D<1$ mm within the 2009 treatment as a whole. Coarse sand $2>D\geq1$ mm was a minor contributor to overall sediment composition. Cobbles $D\geq64$ mm were recovered from the upstream end of the habitat only. The grain-size $D_{50}$ reduced with increased depth, although there was a discernible increase below 25 cm in the midstream core.

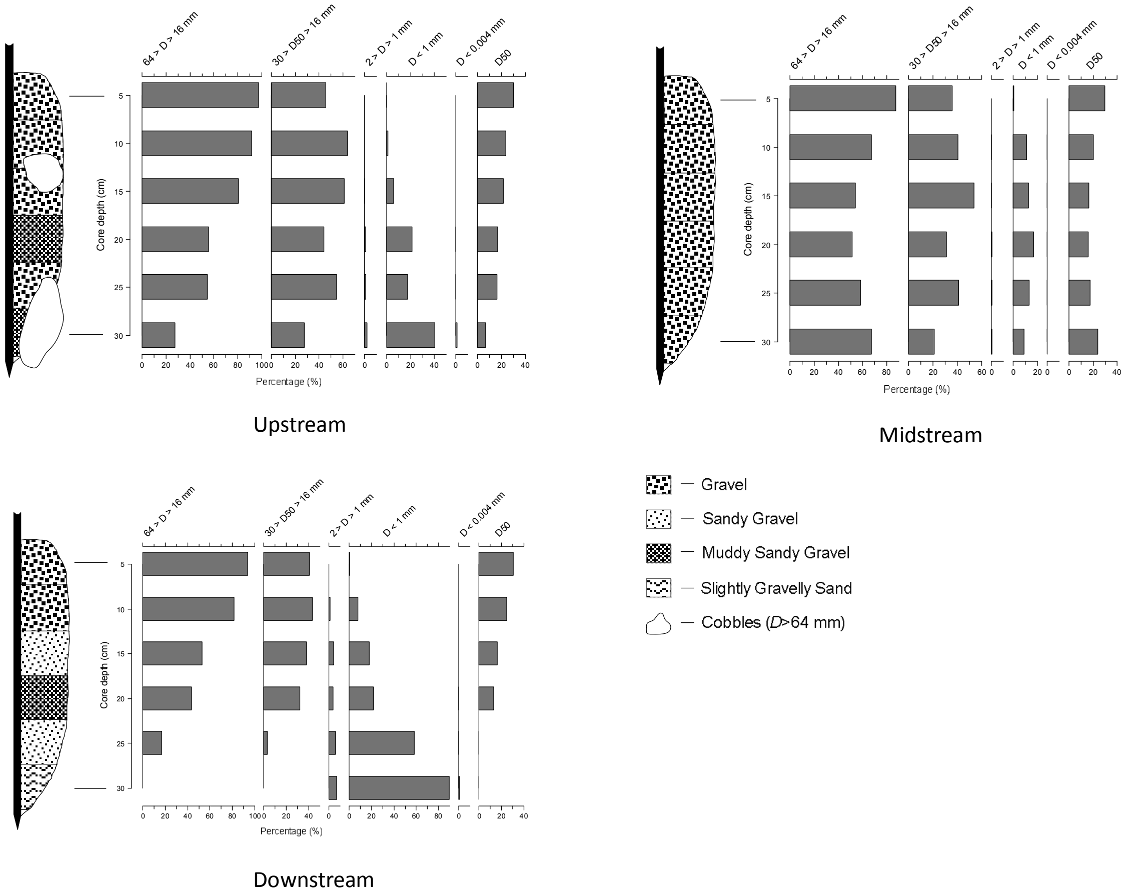



**Figure 4.16** Stratigraphy of rehabilitation gravel site 2009D. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D$≥64 mm were extracted from the upstream core only.

#### 4.3.1.9 2009F

Site 2009F was characterised by high percentages of spawning gravels (64>$D$≥16 mm and 30>$D_{50}$≥16 mm) and small percentages of fine grained sediment (<1 mm), with generally little change in sediment composition throughout the core profile (Figure 4.14 and 4.17). Surface armouring was discernible throughout the site. A slight decrease in $D_{50}$ occurred with depth. Coarse sand 2>$D$≥1 mm was a minor contributor to overall sediment composition. No large clasts $D$≥64 mm were extracted from this site.

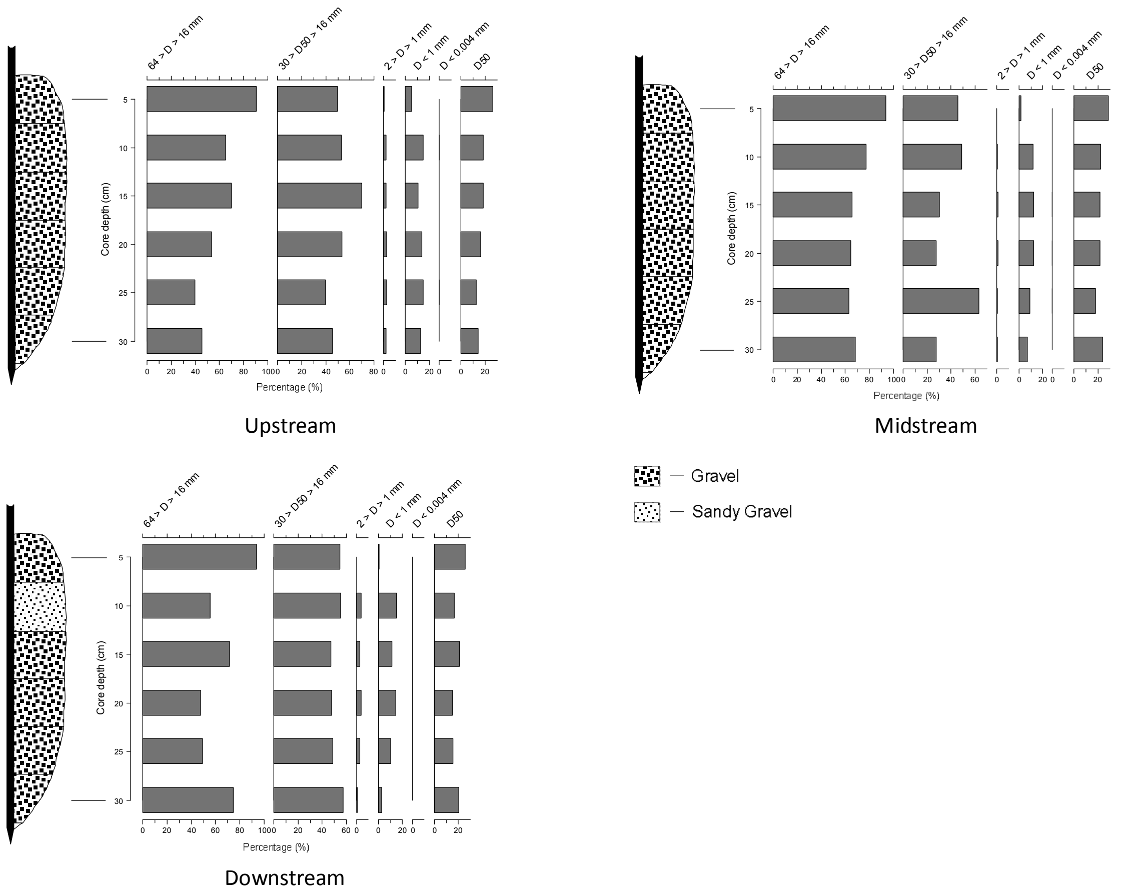



**Figure 4.17** Stratigraphy of the 2009F rehabilitation gravel site. A sediment textual class was assigned to each 5 cm core depth. No cobbles $D$≥64 mm were extracted from the site.

#### 4.3.1.10 2009J

The cumulative percentage grain-size weight was significantly different between cores of the 2009J rehabilitation site (Figure 4.14; Kruskal-Wallis, $p$<0.05, Table 4.2). The grain-size distribution was therefore spatially variable. A high percentage of spawning gravels (64>$D$≥16 mm and 30>$D_{50}$≥16 mm) and low contributions of fine sediment (<1 mm) throughout the mid and upstream areas were a characteristic feature (Figure 4.18). Gravel 64>$D$≥16 mm and 30>$D_{50}$≥16 mm decreased with depth mostly in downstream sediments. Redistribution of gravels suitable for non-migratory *S. trutta* spawning (30>$D_{50}$≥16 mm) were evident throughout the site. Sediment $D$<1 mm increased with depth, particularly in the downstream core, effecting a slight decrease in median grain-size diameter ($D_{50}$) with depth throughout the site. Coarse sand (2>$D$≥1 mm) and clays ($D$<0.004 mm) had minor contributions to sediment composition. Several cobbles ($D$≥64 mm) were present within deeper (15-30 cm) substrate from the midstream and downstream cores of the site. A slight surface armouring was discernible in the downstream core only.

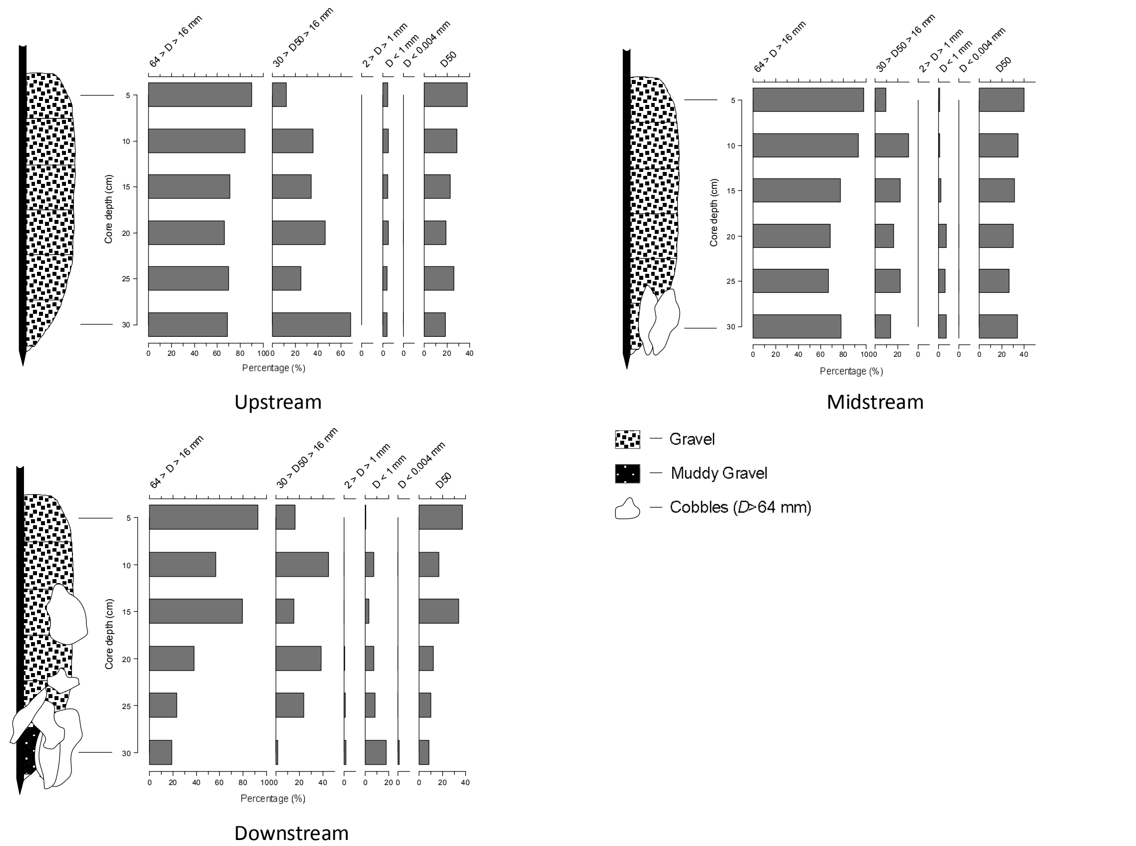



**Figure 4.18** Stratigraphy of rehabilitation gravel site 2009J. A sediment textual class was assigned to each 5 cm core depth. Cobbles $D \geq 64$ mm were extracted from midstream and downstream areas of the site.

## 4.4 Morphosedimentary nature of rehabilitation gravels: evidence from compositional structure and grain-size statistics

Rehabilitation gravel*s*, installed in 2003 and 2009, were constructed to similar specifications (T. Jacklin, pers. comm., 17/01/2011). Therefore the difference in the grain-size distribution between the 2003 rehabilitation gravel and the blueprint grain-size specification provides an indication of how rehabilitation gravel*s* will change in the River Stiffkey. Given a short period of time between the installation of the 2009 rehabilitation gravels and sampling (2011), differences in grain-size distribution between the 2003 and 2009 rehabilitation gravel were used as an indication of longevity.

Cobbles (≥64 mm) constituted the bulk weight of the sampled sediment weighing between 175-4050 g per clast for all cores (Tables 4.4-4.6). The 2009 rehabilitation treatment weight was composed of 46.4% cobbles (≥64 mm), whilst these contributed just 8.9% of the natural substrate weight and 29% of the 2003 treatment weight. A varying degree of roundness from very angular to rounded was observed between cores in each 5 cm increment. Comparatively, natural gravels yielded only angular cobbles (≥64 mm). Given that rehabilitation gravels were installed to similar specifications in 2003 and 2009, sediment erosion was likely responsible for the loss of surface gravels observed in the 2003 rehabilitation gravels. The depth of the overlaying deposit (above the anchoring substrate) was reduced relative to the 2009 rehabilitation gravels; cobbles $D \geq 64$ mm were extracted from deeper positions (23 cm) in the 2009 treatment sites compared to the 2003 rehabilitation gravel (Mann-Whitney, $p<0.05$, Table 4.7), whilst the weighted average depths of natural and 2003 treatments were 13.1 cm and 17.5 cm respectively.

All gravel treatments had unimodal surface grain-size distributions and multimodal subsurface distributions. Median grain-size ($D_{50}$) of all treatments decreased with depth (Figure 4.19, Tables 4.8-4.10). While both the rehabilitation gravel treatments shared similar $D_{50}$ values throughout the vertical extent of the deposit, there was an increase at 20-25 cm depth in the 2003 rehabilitation gravels, likely an indication of the larger substrate deliberately used by the WTT to anchor the deposit during installation. A similar increase was absent in the 2009 rehabilitation gravels; further evidence of surface gravel erosion from the older 2003 rehabilitation gravels. The exposure of anchoring cobbles and small boulders towards the downstream end of each site was a distinct feature of the 2003 gravel treatment. Further, the 2003 rehabilitation gravels had accumulated a greater amount of surface deposition of finer grained sediments than the 2009 rehabilitation gravels. Average $D_{50}$ values of the 2003 and 2009 rehabilitation gravel treatments were 23 mm and 30 mm respectively. $D_{50}$ grain-size of the natural treatment was lower than rehabilitation gravel at all depths of the deposit. Additionally, natural gravels had a much greater extent of surface armouring. A pronounced decrease in $D_{50}$ between surface and subsurface sediments reflected characteristic surface armouring; stratification from a predominantly coarse grained framework to a finer grained matrix (Figure 4.19 and 4.20). Surface armouring was either absent or poorly pronounced in the rehabilitation treatment sites. The gravel structure had little or no stratification consisting of mostly coarse framework gravels and a finer sediment matrix.

Surface sediments of rehabilitation gravel had comparable sorting coefficients as those observed in natural gravels (Figure 4.19). However, with increased depth, natural gravels rapidly became very poorly sorted, unlike the 2009 rehabilitation gravels that had a greater degree of sediment sorting throughout the vertical extent of the deposit. Over time a greater load of finer grained sediment accumulated in interstitial gravel spaces of the 2003 rehabilitation gravels when compared to the 2009 rehabilitation gravels. The older 2003 rehabilitation gravels were less well sorted with increased depth. Coarse grained sediments were an important part of the grain-size distribution for each 5 cm increment of rehabilitation and natural gravel treatments. All treatments had strong fine skewed distributions, however rehabilitation gravel became more finely skewed with increased depth (Figure 4.19). Grain-size distributions were very platykurtic to very leptokurtic for the 2009 treatment, platykurtic to leptokurtic for the 2003 treatment and very platykurtic to leptokurtic for the natural treatment (Figure 4.19). Natural gravels had a high composition of small chalk aggregates (Figure 4.20) and contained greater proportions of smaller grained sediments throughout the deposit than either of the rehabilitation gravel treatments.

**Table 4.4** Summary details of cobbles $D \geq 64$ mm sampled from natural treatment gravels. Axis dimensions are in mm. The core and depth of each cobble as well as axes measurements, roundness descriptions and weight are noted. The percentage weight that each cobble contributed to the core weight was summarised as well as the cumulative cobble contribution to the core, site and treatment weight (%). Percentage (%) weight contribution of cobbles $D \geq 64$ mm to the natural gravel treatment was low, just 8.9%.

| Site | Core | Depth | Axis | | | Roundness | Weight (g) | $D \geq 64$ mm (%) | Core (%) | Site (%) | Treatment (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | a | b | c | | | | | | |
| Fort | 1 | 5-15 | 166 | 90 | 59 | angular | 625 | 18.9 | 18.9 | | |
| | 2 | 10-20 | 85 | 80 | 61 | angular | 175 | 3.5 | | | |
| | | 10-25 | 132 | 110 | 80 | angular | 1550 | 31.4 | 35.0 | 15.1 | |
| W.Hall | 2 | 5-15 | 108 | 60 | 45 | very angular | 431.8 | 8.2 | 8.2 | 3.6 | 8.9 |

**Table 4.5** Summary details of cobbles $D \geq 64$ mm sampled from the 2003 rehabilitation gravel treatment. Axis dimensions are in mm. The core and depth of each cobble as well as axes measurements, roundness descriptions and weight are noted. The percentage weight that each cobble contributed to the core weight was summarised as well as the cumulative cobble contribution to the core, site and treatment weight (%). A large percentage (%) weight of sediment sampled from site 2003B and 2003C was comprised of cobbles $D \geq 64$ mm, 40.8% and 65.6% respectively. The average for the treatment was high, 45.3%.

| Site | Core | Depth | Axis | | | Roundness | Weight (g) | $D \geq 64$ mm (%) | Core (%) | Site (%) | Treatment (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | a | b | c | | | | | | |
| 2003 A | 2 | 0-20 | 147 | 87 | 69 | sub-rounded | 1150 | 23.9 | 23.9 | 11.2 | |
| 2003 B | 1 | 0-10 | 116 | 110 | 75 | rounded | 1425 | 16.9 | | | |
| | | 10-20 | 110 | 111 | 70 | sub-angular | 1125 | 13.4 | | | |
| | | 15-30 | 140 | 85 | 50 | angular | 600 | 7.1 | | | |
| | | 20-30 | 135 | 125 | 56 | sub-rounded | 1200 | 14.3 | 51.7 | | |
| | 3 | 15-25 | 147 | 130 | 67 | sub-rounded | 2200 | 41.0 | 41.0 | 40.8 | |
| 2003 C | 1 | 0-10 | 175 | 160 | 150 | sub-rounded | 3100 | 29.6 | | | |
| | | 5-15 | 123 | 105 | 41 | rounded | 801 | 7.7 | | | |
| | | 5-15 | 110 | 100 | 90 | sub-rounded | 1479 | 14.1 | | | |
| | | 20-30 | 120 | 75 | 70 | angular | 1055 | 10.1 | 61.5 | | |
| | 2 | 10-20 | 90 | 65 | 54 | sub-rounded | 365 | 5.1 | | | |
| | | 10-20 | 13.5 | 130 | 75 | sub-rounded | 2150 | 30.0 | | | |
| | | 10-20 | 100 | 87 | 50 | angular | 400 | 5.6 | | | |
| | | 15-25 | 130 | 65 | 49 | sub-angular | 425 | 5.9 | | | |
| | | 20-30 | 113 | 58 | 54 | angular | 200 | 2.8 | | | |
| | | 20-30 | 150 | 110 | 68 | sub-rounded | 1300 | 18.2 | | | |
| | | 20-30 | 115 | 80 | 41 | very angular | 300 | 4.2 | 71.8 | | |
| | 3 | 20-35 | 185 | 105 | 80 | angular | 2050 | 64.9 | 64.9 | 65.6 | 45.3 |

**Table 4.6** Summary details of cobbles $D \geq 64$ mm sampled from the 2009 rehabilitation gravel treatment. Axis dimensions are in mm. The core and depth of each cobble as well as axes measurements, roundness descriptions and weight are noted. The percentage weight that each cobble contributed to the core weight was summarised as well as the cumulative cobble contribution to the core, site and treatment weight (%). Cobbles $D \geq 64$ mm had a high percentage (%) weight contribution to the 2009 rehabilitation gravel treatment, 46.4%.

| Site | Core | Depth | Axis | | | Roundness | Weight (g) | $D \geq 64$ mm% | Core% | Site% | Treatment% |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | a | b | c | | | | | | |
| 2009A | 2 | 10-25 | 250 | 139 | 70 | sub-rounded | 4050 | 31.7 | | | |
| | | 20-35 | 122 | 95 | 65 | rounded | 1050 | 8.2 | | | |
| | | 25-35 | 150 | 99 | 71 | angular | 1075 | 8.4 | | | |
| | | 10-25 | 167 | 121 | 94 | angular | 2600 | 20.3 | 68.6 | 15.1 | |
| 2009D | 1 | 5-15 | 130 | 114 | 40 | sub-rounded | 756.1 | 11.6 | | | |
| | | 20-35 | 176 | 110 | 108 | sub-rounded | 3100 | 47.5 | 59.1 | 25.8 | |
| 2009J | 2 | 20-35 | 147 | 140 | 95 | angular | 1825 | 29.0 | | | |
| | | 20-35 | 128 | 86 | 75 | sub-angular | 1100 | 17.5 | 46.5 | | |
| | 3 | 5-20 | 140 | 123 | 80 | angular | 1525 | 13.9 | | | |
| | | 20-35 | 135 | 111 | 82 | rounded | 2050 | 18.7 | | | |
| | | 20-30 | 100 | 75 | 32 | sub-angular | 332.4 | 3.0 | | | |
| | | 20-35 | 120 | 120 | 120 | sub-rounded | 1281.7 | 11.7 | | | |
| | | 15-25 | 90 | 70 | 40 | rounded | 440.8 | 4.0 | | | |
| | | 20-30 | 140 | 115 | 80 | sub-rounded | 1850 | 16.8 | 68.1 | 9.1 | 46.4 |

**Table 4.7** Summary of the statistical test results of difference in the depth of cobbles (*D*>64) mm between gravel treatments.

| | **Kruskal-Wallis** | **Mann-Whitney U** | |
|---|---|---|---|
| | | **2003** | **2009** |
| **Treatment** | 1 | - | - |
| **Natural** | - | 0 | 1 |
| **2003** | - | - | 1 |

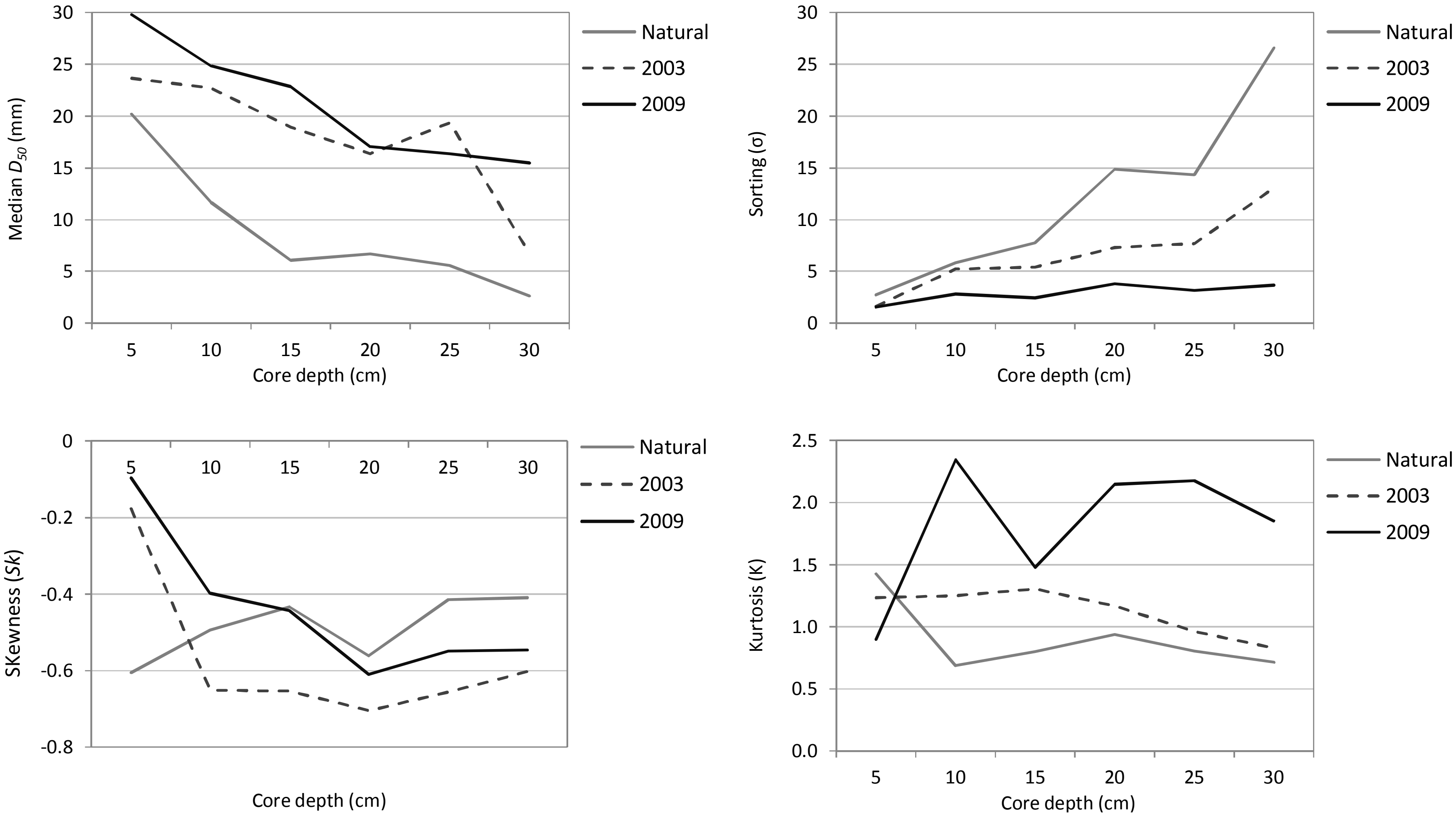



**Figure 4.19** Grain-size distribution sample statistics for each treatment based on the Folk and Ward (1957) method for the median of each 5 cm core level from all core samples within each treatment. Natural gravel had a lower $D_{50}$ throughout the core profile than either rehabilitation gravel treatment. This is likely due to the contribution of a low percentage of gravel (64>*D*≥16 mm) and high fine sediment (<1 mm). Both the 2003 and 2009 rehabilitation gravel treatments had well sorted sediment compositions throughout the core profile, reflecting the greater percentage of gravel (64>*D*≥16 mm) relative to natural gravels.

**Table 4.8** Summary percentile statistics for the natural gravel treatment. Median ($D_{50}$) grain-sizes decreased with depth.

| | | Fort (mm) | | | W.Hall (mm) | | | W.Curd (mm) | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Depth (cm) | | $D_{10}$ | $D_{50}$ | $D_{90}$ | $D_{10}$ | $D_{50}$ | $D_{90}$ | $D_{10}$ | $D_{50}$ | $D_{90}$ |
| Core 1 | 0-5 | 4.76 | 20.22 | 31.43 | 0.57 | 12.02 | 25.61 | 0.42 | 12.86 | 28.65 |
| | 5-10 | 0.38 | 13.62 | 49.81 | 0.43 | 7.17 | 21.41 | 3.56 | 17.89 | 76.82 |
| | 10-15 | 0.02 | 3.59 | 31.75 | 0.33 | 6.07 | 21.12 | 0.30 | 6.47 | 23.15 |
| | 15-20 | 0.004 | 9.47 | 46.27 | 0.32 | 10.79 | 46.16 | 0.06 | 1.02 | 20.45 |
| | 20-25 | 0.004 | 9.16 | 44.34 | 0.29 | 5.58 | 25.31 | 0.004 | 0.42 | 5.44 |
| | 25-30 | 0.003 | 3.50 | 15.84 | 0.05 | 2.64 | 15.17 | 0.004 | 0.32 | 4.14 |
| Core 2 | 0-5 | 4.06 | 22.90 | 49.31 | 0.68 | 18.68 | 31.60 | 9.03 | 35.94 | 54.16 |
| | 5-10 | 0.35 | 4.64 | 19.88 | 0.40 | 9.95 | 34.14 | 0.52 | 22.85 | 50.84 |
| | 10-15 | 0.17 | 2.74 | 23.49 | 0.32 | 7.89 | 26.73 | 0.20 | 8.37 | 44.81 |
| | 15-20 | 0.004 | 9.47 | 46.27 | 0.22 | 7.46 | 25.42 | 0.01 | 6.71 | 22.34 |
| | 20-25 | 0.23 | 6.97 | 41.84 | 0.36 | 10.97 | 40.42 | 0.003 | 0.42 | 17.00 |
| | 25-30 | 0.003 | 6.65 | 42.53 | 0.33 | 8.26 | 34.13 | 0.003 | 0.06 | 13.97 |
| Core 3 | 0-5 | 0.01 | 10.71 | 28.85 | 9.10 | 20.38 | 38.99 | 11.40 | 23.15 | 45.40 |
| | 5-10 | 0.002 | 1.08 | 19.15 | 0.57 | 11.65 | 25.14 | 0.52 | 16.97 | 49.80 |
| | 10-15 | 0.002 | 2.45 | 21.88 | 0.50 | 10.00 | 47.44 | 0.13 | 3.19 | 13.46 |
| | 15-20 | 0.002 | 5.08 | 24.31 | 0.38 | 4.18 | 23.77 | 0.003 | 0.36 | 9.16 |
| | 20-25 | 0.002 | 5.46 | 44.71 | 0.42 | 10.54 | 42.07 | 0.003 | 0.32 | 16.52 |
| | 25-30 | 0.002 | 1.63 | 19.75 | 0.41 | 9.01 | 24.27 | 0.003 | 0.31 | 14.82 |

**Table 4.9** Summary percentile statistics for the 2003 rehabilitation gravel treatment. Sediment increments between 20-30 cm depth were not sampled for site 2003B due to a core blockage.

| | | 2003A (mm) | | | 2003B (mm) | | | 2003C (mm) | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Depth (cm) | | ***D*$_{10}$** | ***D*$_{50}$** | ***D*$_{90}$** | ***D*$_{10}$** | ***D*$_{50}$** | ***D*$_{90}$** | ***D*$_{10}$** | ***D*$_{50}$** | ***D*$_{90}$** |
| Core 1 | 0-5 | 0.02 | 0.04 | 0.05 | 11.39 | 23.64 | 47.22 | 0.57 | 22.73 | 47.10 |
| | 5-10 | 8.54 | 27.46 | 51.28 | 0.55 | 23.75 | 50.35 | 0.51 | 19.51 | 44.55 |
| | 10-15 | 2.54 | 25.60 | 50.63 | 0.20 | 18.02 | 48.88 | 0.51 | 18.91 | 37.29 |
| | 15-20 | 0.39 | 15.57 | 40.22 | 0.15 | 35.24 | 53.94 | 0.35 | 18.78 | 35.67 |
| | 20-25 | 1.81 | 32.35 | 76.18 | 0.04 | 3.62 | 46.64 | 0.24 | 19.28 | 42.71 |
| | 25-30 | 0.08 | 30.98 | 52.57 | 0.02 | 0.97 | 24.93 | 0.02 | 4.88 | 25.69 |
| Core 2 | 0-5 | 24.06 | 67.54 | 77.33 | 15.99 | 23.68 | 45.91 | 10.48 | 22.82 | 44.64 |
| | 5-10 | 18.75 | 37.84 | 54.71 | 7.10 | 22.93 | 46.76 | 0.29 | 10.94 | 25.25 |
| | 10-15 | 0.73 | 37.97 | 54.75 | 0.69 | 21.35 | 46.09 | 0.44 | 28.99 | 52.11 |
| | 15-20 | 0.27 | 20.74 | 43.21 | 0.80 | 19.86 | 27.62 | 0.05 | 9.46 | 37.01 |
| | 20-25 | 0.48 | 37.43 | 54.60 | - | - | - | 0.02 | 0.78 | 12.24 |
| | 25-30 | 0.57 | 58.82 | 75.46 | - | - | - | 0.04 | 8.57 | 25.38 |
| Core 3 | 0-5 | 16.81 | 29.57 | 52.08 | 17.61 | 35.02 | 53.87 | 14.21 | 38.01 | 54.76 |
| | 5-10 | 0.30 | 16.94 | 26.76 | 10.46 | 22.33 | 41.25 | 0.54 | 22.69 | 48.84 |
| | 10-15 | 0.29 | 10.94 | 24.94 | 0.81 | 17.01 | 26.78 | 0.38 | 13.05 | 26.15 |
| | 15-20 | 0.19 | 12.68 | 25.95 | 0.51 | 16.36 | 28.99 | 0.02 | 1.52 | 23.24 |
| | 20-25 | 1.39 | 19.46 | 27.51 | 3.99 | 37.55 | 54.63 | 0.01 | 0.64 | 15.93 |
| | 25-30 | 0.15 | 20.53 | 42.03 | 0.02 | 3.66 | 21.88 | 0.01 | 0.59 | 11.15 |

**Table 4.10** Summary percentile statistics for the 2009 rehabilitation gravel treatment.

| | | 2009A (mm) | | | 2009D (mm) | | | 2009F (mm) | | | 2009J (mm) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Depth (cm)** | $D_{10}$ | $D_{50}$ | $D_{90}$ | $D_{10}$ | $D_{50}$ | $D_{90}$ | $D_{10}$ | $D_{50}$ | $D_{90}$ | $D_{10}$ | $D_{50}$ | $D_{90}$ |
| Core 1 | 0-5 | 17.12 | 24.30 | 45.93 | 17.75 | 30.53 | 52.42 | 16.15 | 26.69 | 50.60 | 16.24 | 38.39 | 54.87 |
| | 5-10 | 17.01 | 31.85 | 52.86 | 16.34 | 24.22 | 46.93 | 0.59 | 19.18 | 33.64 | 10.52 | 29.21 | 52.01 |
| | 10-15 | 12.12 | 23.69 | 47.03 | 9.08 | 21.93 | 41.95 | 0.84 | 19.21 | 27.44 | 8.49 | 23.53 | 49.65 |
| | 15-20 | 8.02 | 19.08 | 27.40 | 0.39 | 17.37 | 32.17 | 0.65 | 16.73 | 26.69 | 8.14 | 19.98 | 42.20 |
| | 20-25 | 0.59 | 13.15 | 25.49 | 0.52 | 16.94 | 26.76 | 0.59 | 13.14 | 25.64 | 8.67 | 26.56 | 51.45 |
| | 25-30 | 0.50 | 15.48 | 26.38 | 0.04 | 7.21 | 23.97 | 0.66 | 14.59 | 26.18 | 8.24 | 19.04 | 27.39 |
| Core 2 | 0-5 | 8.82 | 21.63 | 47.83 | 14.07 | 30.66 | 52.46 | 16.89 | 29.06 | 51.88 | 27.82 | 40.62 | 55.50 |
| | 5-10 | 9.22 | 62.38 | 76.12 | 0.57 | 20.94 | 46.25 | 0.65 | 22.82 | 47.05 | 17.37 | 34.93 | 53.85 |
| | 10-15 | 9.38 | 61.62 | 75.93 | 0.48 | 16.84 | 26.73 | 0.64 | 22.18 | 49.28 | 10.26 | 32.06 | 52.93 |
| | 15-20 | 0.19 | 13.11 | 25.70 | 0.32 | 16.60 | 42.66 | 0.65 | 22.35 | 49.69 | 8.45 | 30.85 | 52.53 |
| | 20-25 | 2.83 | 62.20 | 76.07 | 0.52 | 18.26 | 40.17 | 1.21 | 18.28 | 27.17 | 8.38 | 27.10 | 51.73 |
| | 25-30 | 0.51 | 13.77 | 39.04 | 1.22 | 24.42 | 50.72 | 4.39 | 24.36 | 50.59 | 8.48 | 34.81 | 53.81 |
| Core 3 | 0-5 | 17.19 | 25.26 | 48.36 | 17.06 | 31.29 | 52.67 | 16.75 | 26.40 | 50.17 | 17.87 | 38.05 | 54.78 |
| | 5-10 | 8.42 | 27.60 | 51.39 | 2.18 | 25.40 | 50.11 | 0.64 | 17.10 | 26.81 | 5.42 | 17.66 | 33.17 |
| | 10-15 | 12.12 | 23.69 | 47.03 | 0.60 | 16.87 | 37.68 | 0.82 | 21.29 | 44.93 | 8.40 | 34.93 | 53.85 |
| | 15-20 | 8.02 | 19.08 | 27.40 | 0.51 | 13.48 | 32.12 | 0.66 | 15.28 | 26.34 | 2.72 | 12.93 | 25.51 |
| | 20-25 | 0.59 | 13.15 | 25.49 | 0.26 | 0.73 | 36.64 | 0.92 | 15.83 | 26.42 | 1.64 | 10.63 | 23.10 |
| | 25-30 | 0.50 | 15.48 | 26.38 | 0.22 | 0.55 | 1.01 | 8.79 | 21.06 | 40.35 | 0.05 | 9.04 | 40.49 |

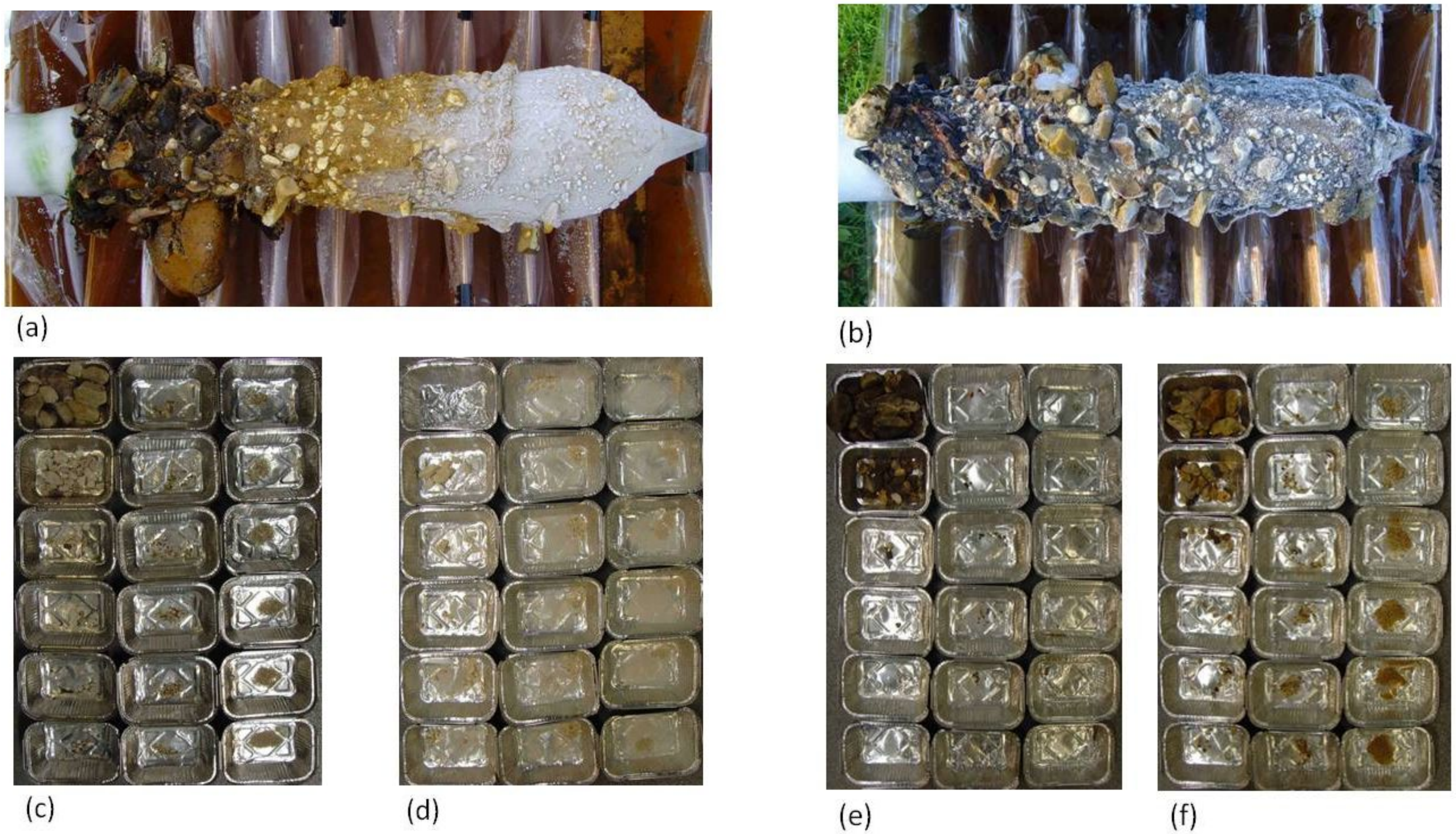



**Figure 4.20** Comparison of a typical core (core 2, Water Hall) from a natural treatment site (a) and a rehabilitation treatment site (core 1, 2009J) (b), and the classification of sediment sizes from surface substrate (c) and (e) and from deeper substrate (d) and (f) of the core after sieving. Note the abundance of larger sediments in the upper core of natural gravel deposits and their absence in the deeper portion, whilst rehabilitation gravels (b), (e) and (f) had an abundance gravel ($64>D\geq16$ mm) throughout the core profile.

### 4.5 Sediment composition: a comparative analysis

Multivariate analysis using ordination techniques was used to describe the sediment composition of rehabilitation gravels. Percentage textural classification descriptors, analysed in GRADISTAT v4 (Blott and Pye, 2001), of the cumulative core grain-size distributions of each site, consistent with Cefas (1999), were determined. These were combined into gravel (64>*D*≥2 mm), sand (2>*D*≥0.063 mm) and silt (*D*<0.0063) sediment size-class descriptors consistent with Wentworth (1922). Gradients of sediment data were established along axes that were not constrained by environmental data and as such provided sedimentological differences between sites.

**Table 4.11** Summary of the Detrended Correspondence Analysis (DCA) of sediment sampled at each site within all treatments and each treatment individually.

| Ordination | Axes | 1 | 2 | 3 | 4 | Total inertia |
|---|---|---|---|---|---|---|
| Treatments | Eigenvalues | 0.135 | 0.016 | 0 | 0 | 0.173 |
| | Lengths of gradient | 0.74 | 0.486 | 0 | 0 | |
| | Cumulative percentage variance | 77.9 | 87 | 0 | 0 | |
| Natural | Eigenvalues | 0.066 | 0.012 | 0 | 0 | 0.086 |
| | Lengths of gradient | 0.558 | 0.291 | 0 | 0 | |
| | Cumulative percentage variance | 76.5 | 89.9 | 0 | 0 | |
| 2003 | Eigenvalues | 0.012 | 0.001 | 0 | 0 | 0.014 |
| | Lengths of gradient | 0.19 | 0.062 | 0 | 0 | |
| | Cumulative percentage variance | 88.6 | 95.1 | 0 | 0 | |
| 2009 | Eigenvalues | 0.068 | 0.001 | 0 | 0 | 0.071 |
| | Lengths of gradient | 0.411 | 0.071 | 0 | 0 | |
| | Cumulative percentage variance | 96.8 | 98.7 | 0 | 0 | |

Detrended correspondence analysis (DCA) of percentage (%) gravel (64>*D*≥2 mm), sand (2>*D*≥0.063 mm) and silt (*D*<0.0063 mm) for each site and treatment type gave a gradient length <3 standard deviation units for Axis I (Table 4.11). Given this relatively short gradient, principal components analysis (PCA) was considered the most appropriate method for further analyses. Sediment variance of rehabilitation gravels was driven by a gravel-sand gradient along component axis 1 accounting for 96.7% and 99.4% of data variation for 2003 and 2009 treatment sites respectively (Table 4.12; Figure 4.21a and b). The second principal component increased with composition of silt/clay and explained a further 3% and 0.6% variation for 2003 and 2009 treatment sites respectively. Axis 1 and 2 explained >99% variation for each treatment. Unlike natural gravels, neither the 2003 nor the 2009 rehabilitation gravels were compositionally unique and illustrated similar grain-size characteristics along a large gravel-sand gradient. Natural gravels at control sites, Water Hall, Fort and Whey Curd, consisted of a broad size-range of sediments with each site compositionally distinct (Figure 4.21c); the first principal component axis was correlated by a gravel-silt gradient accounting for 66.6% of the variation (Table 4.12). Decreasing composition of sand, along principal component axis 2, explained 33.3% of data variation. Substrate at the Water Hall site were dominated by sand and were compositionally similar but varied along a gravel-sand gradient. The Fort site substrate had less similarity with a large gravel-silt/clay gradient range. Sediments at the Whey Curd site varied over a large sand-silt/clay gradient. Natural sites Fort and Whey Curd had the lowest gravel composition. Axes 1 and 2 cumulatively explained 99.9% of the variation in the treatment data (Table 4.12). The first principal component was closely correlated with percentage gravel (Figure 4.21d). Rehabilitation gravels had a greater composition of gravel than natural sites. Axis 2 was positively correlated with percentage silt and negatively with percentage sand. Rehabilitation gravel sites were generally grouped in ordination space, except 3d3 (2003D, core 3), and illustrated a similar sediment grain-size composition.

**Table 4.12** Summary table of variance described by each axis for the PCA ordination. Axes 1 and 2 account for ≥99.9% variation within the data for each of the analyses. See the associated PCA biplot (Figure 4.21) for sediment composition.

| Biplot | Axes | 1 | 2 | 3 | 4 | Total variance |
|---|---|---|---|---|---|---|
| Treatments | Eigenvalues | 0.874 | 0.126 | 0.001 | 0 | 1 |
| | Cumulative percentage variance | 87.4 | 99.9 | 100 | 0 | |
| Natural | Eigenvalues | 0.666 | 0.333 | 0 | 0 | 1 |
| | Cumulative percentage variance | 66.6 | 100 | 100 | 0 | |
| 2003 | Eigenvalues | 0.967 | 0.033 | 0 | 0 | 1 |
| | Cumulative percentage variance | 96.7 | 100 | 100 | 0 | |
| 2009 | Eigenvalues | 0.994 | 0.006 | 0.001 | 0 | 1 |
| | Cumulative percentage variance | 99.4 | 99.9 | 100 | 0 | |

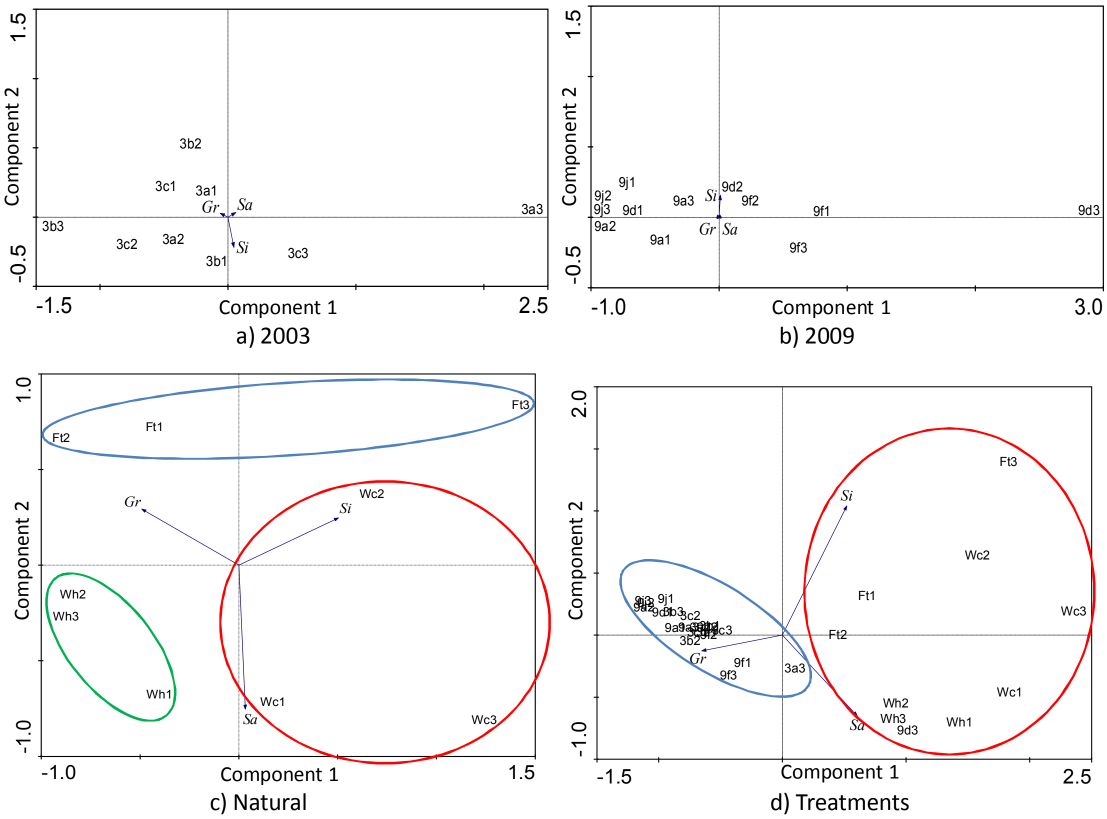



**Figure 4.21** Principal components analysis (PCA) biplot of the cumulative sediment distribution: gravel (Gr), sand (Sa) and silt/clay (Si) from each site and treatment. The first text pair of sample codes represents the site and the number refers to a sample specific to that site. For example, '*3a2*' is the second core sampled at site 2003A. Natural sites are coded '*Ft*' for Fort, '*Wc*' for Whey Curd and '*Wh*' for Water Hall. Note the sediment grain-size variance within the natural treatment sites and the similarity of sediments within the rehabilitation gravel treatments. See PCA ordination variance statistics summary Table 4.12. See the site map (Figure 2.1, Chapter 2) for the location of each gravel site.

### 4.6 Association between stream velocity and surface sediments

Water velocities were low across all gravel treatments over the duration of the study (<1.0 m $s^{-1}$), with most velocities within the range 0.1-0.4 m $s^{-1}$ each year (Figure 4.22). Froude numbers were never greater than 0.25 (site 2009G, year 2011), indicating that velocity remained slow for all sites during all years. The rehabilitation gravels were not associated with increased stream velocities. Apart from the 2009 and natural gravel treatment in 2010 ($Chi^2$, $p<0.05$, Table 4.13), intra-annual mean gravel treatment water velocities were relatively similar, with the 2003 rehabilitation gravels and natural gravels more comparable. However, the 2009 rehabilitation gravels consistently had a wider range of velocities, but not the highest mean. Significantly greater variability was observed between non-consecutive years (2010 and 2012) for the natural and 2009 treatments, although water velocities remained sub critical ($Chi^2$, $p<0.05$, Table 4.14).

Stream velocities were recorded over mostly 20-50 cm water depth (Figure 4.23), between July and September in 2010, 2011 and 2012. Mean water depths for the 2003 and 2009 rehabilitation gravels were 35.2 cm and 38.8 cm respectively. Although the mean natural gravel depth was slightly lower than recorded for the rehabilitation gravels (30.4 cm), natural gravels had a greater degree of interannual consistency (29-32 cm). Rehabilitation gravel stream velocities were recorded over a wide range of depths, with mean depths ranging between 30-41.5 cm, due to the ingot-like nature of the gravel features with steep up and downstream extremities (see Figure 3.17, Chapter 3). This characteristic was a distinctive rehabilitation gravel feature and not observed across the natural gravels.

Ordination methods were used to investigate the effect of velocity on sediment composition between sites. Mean velocity, measured at 5 cm above the stream-bed of each site over three years, was used to investigate the relationship with the surface 0-5 cm of percent gravel ($64>D\geq2$ mm), sand ($2>D\geq0.063$ mm) and silt ($D<0.0063$ mm), consistent with the grade scale classes of Wentworth (1922). DCA determined a gradient length <3 standard deviation (SD) units for Axis I for all ordination analyses (Table 4.15). Based on the short gradient principal components analysis (PCA) was the more appropriate technique for further analyses. Axes 1 and 2 always cumulatively explained 82% of the variation in the data. Principal components axis 1 was correlated with the relationship between surface (0-5 cm) gravel and velocity, accounting for 78.9% of the observed variance (Table 4.16; Figure 4.24).

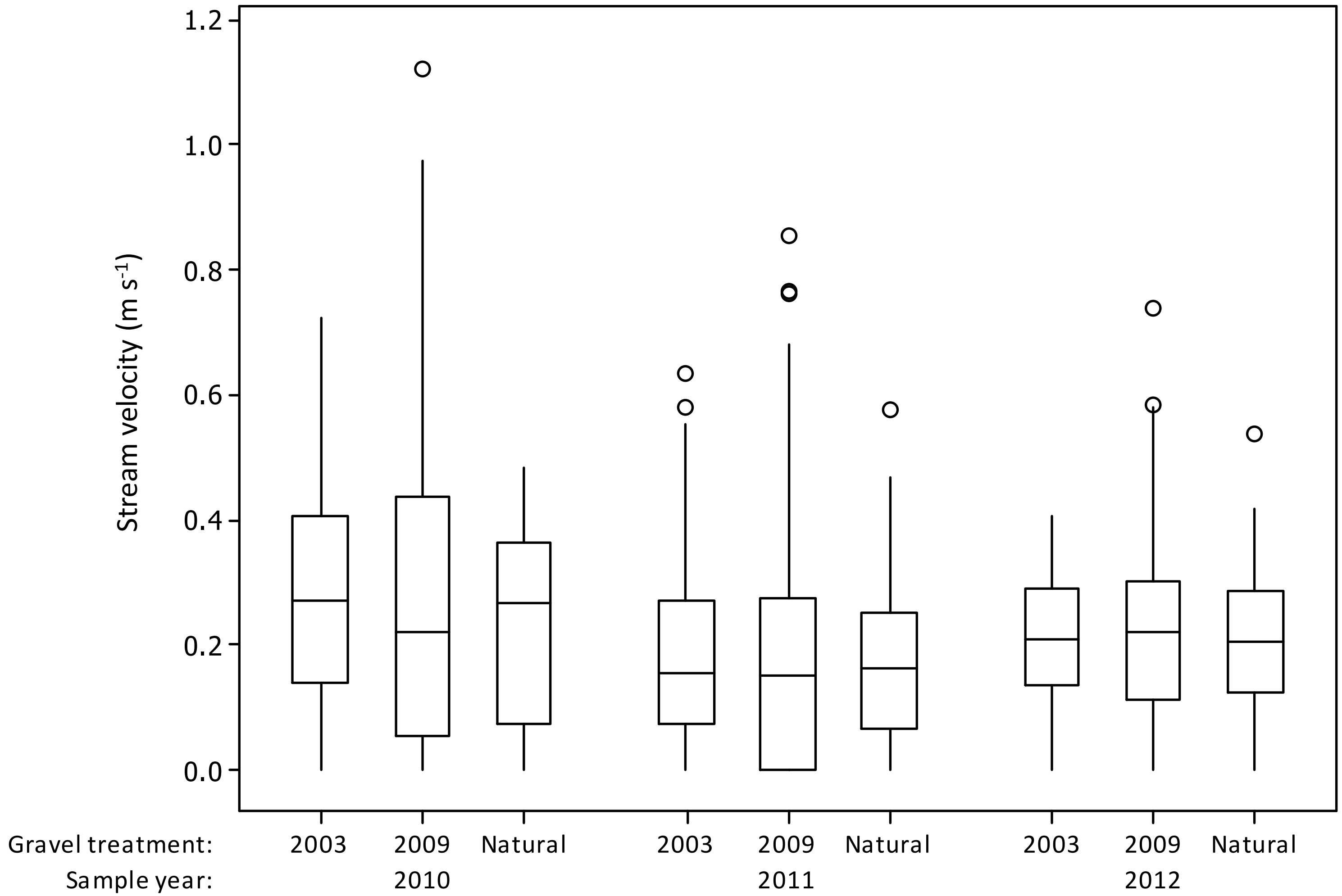



**Figure 4.22** Mean water velocity box plot for gravel treatments per year recorded at each site between July and September 2010 to 2012. Water velocities were mostly low during the summer months (<1.0 m $s^{-1}$).

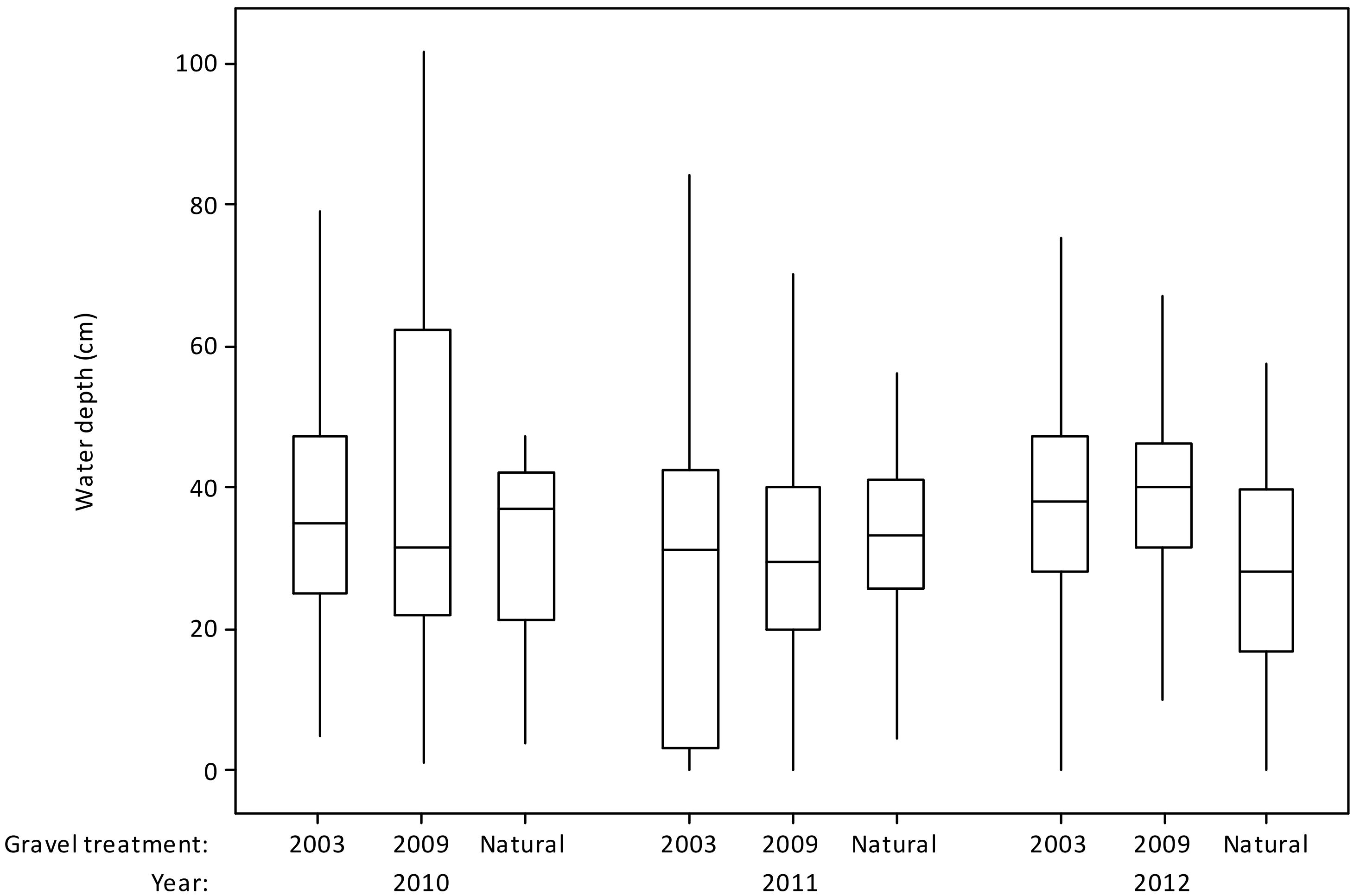



**Figure 4.23** Boxplot of mean water depth (cm) recorded at each site within each gravel treatment between July and September 2010 to 2012. The mean of each gravel treatment is given for each year. Outliers have been removed from the plot to expand the *y*-scale of the main data.

**Table 4.13** Summary results of the $Chi^2$ test of difference in velocity between gravel treatments within each study year.

| | | 2003 Treatment | | | Natural Treatment | | |
|---|---|---|---|---|---|---|---|
| **Year** | **Treatment** | $\chi^2$ | **d.f** | **p-value** | $\chi^2$ | **d.f** | **p-value** |
| 2010 | 2009 | 10.433 | 8 | 0.236 | 15.649 | 8 | 0.048 |
| | Natural | 11.195 | 8 | 0.191 | - | - | - |
| 2011 | 2009 | 4.823 | 8 | 0.776 | 5.772 | 8 | 0.673 |
| | Natural | 4.367 | 8 | 0.823 | - | - | - |
| 2012 | 2009 | 3.687 | 8 | 0.884 | 6.007 | 8 | 0.646 |
| | Natural | 1.544 | 8 | 0.992 | - | - | - |

**Table 4.14** Summary results of $Chi^2$ tests for difference in velocity within each of the gravel treatments over the study period, 2010-2012.

| | | 2011 | | | 2012 | | |
|---|---|---|---|---|---|---|---|
| | **Year** | $\chi^2$ | **d.f** | **p-value** | $\chi^2$ | **d.f** | **p-value** |
| **Natural** | 2010 | 15.144 | 8 | 0.056 | 19.765 | 8 | 0.011 |
| | 2011 | - | - | - | 6.131 | 8 | 0.633 |
| **2003** | 2010 | 22.852 | 8 | 0.004 | 21.349 | 8 | 0.006 |
| | 2011 | - | - | - | 7.937 | 8 | 0.440 |
| **2009** | 2010 | 11.665 | 8 | 0.167 | 18.853 | 8 | 0.016 |
| | 2011 | - | - | - | 7.671 | 8 | 0.466 |

**Table 4.15** Summary Detrended Correspondence Analysis (DCA) results of the relationship between stream velocity and surface substrate.

| **Axes** | **1** | **2** | **3** | **4** | **Total inertia** |
|---|---|---|---|---|---|
| Eigenvalues | 0.07 | 0.003 | 0.001 | 0 | 0.088 |
| Lengths of gradient | 0.547 | 0.153 | 0.357 | 0 | |
| Cumulative percentage variance | 79.2 | 82.4 | 83.2 | 0 | |

**Table 4.16** Summary table of the variance described by each axis for the PCA ordination. See the associated PCA biplot (Figure 4.24) for the relationship between sediment composition and velocity.

| **Axes** | **1** | **2** | **3** | **4** | **Total variance** |
|---|---|---|---|---|---|
| Eigenvalues | 0.789 | 0.138 | 0.07 | 0.002 | 1 |
| Cumulative percentage variance | 78.9 | 92.7 | 99.8 | 100 | |

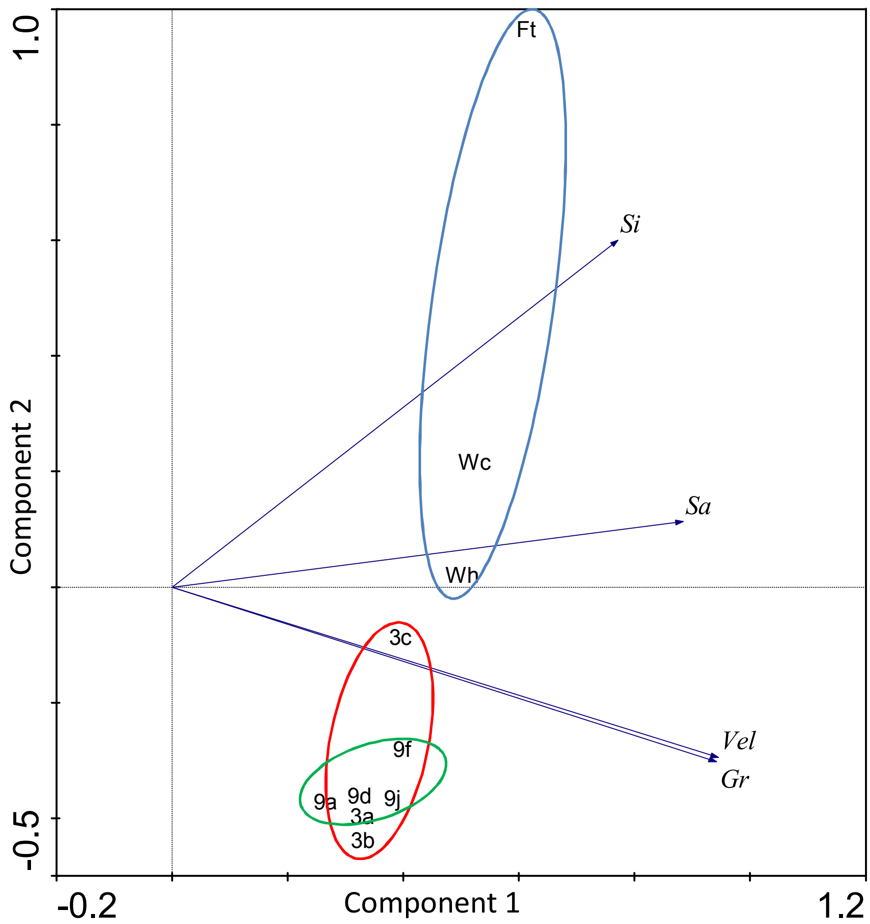



**Figure 4.24** Principal components analysis (PCA) biplot of a cumulative classified distribution of the surface 5 cm of river bed sediment; gravel (Gr), sand (Sa) and silt/clay (Si), and mean velocity (Vel) measured at each site. PCA statistics are summarised in Table 4.16. Each code represents a specific site. For example, '*3a*' is site 2003A, and '*Wh*' is the Water Hall site.

Composition of silt explained 13.8% variation along axis 2; collectively this PCA accounted for 92.7% of the variance in the data. Rehabilitation gravel sites illustrated further morphological similarity as surface gravels were correlated with velocity. Given that rehabilitation gravels had a large percentage of gravel, this relationship with velocity was expected. Site 2003C (*3c*) however had a lower gravel composition and had a greater association with natural gravels. Natural gravel sites had lower velocities and correspondingly greater silt and sand compositions. However, those sites with greater velocities had an associated gravel composition. Within the natural gravel treatment the Water Hall site had the highest velocities and gravel content whilst the Fort site had a very poor velocity and the least percentage of gravel. Moreover, silt and clay sized sediments were more abundant at the Fort site.

### 4.6.1 Fine grained sediment (<1 mm) accrual within the embryo incubation zone, 5-20 cm

*S. trutta* excavate a pit 5-20 cm deep in which to deposit eggs during the redd cutting process (Crisp and Carling, 1989). As such fine grained sediment (<1 mm) accrual within this incubation zone has greatest impact on the development of *S. trutta* embryos. The cumulative percentage sediment $D$<1 mm weight varied significantly between gravel treatments within the embryo incubation zone (Man-Whitney, $p$<0.05, Table 4.17). The natural gravel treatment had a significantly greater cumulative sediment $D$<1 mm percentage than either of the rehabilitation gravel treatments (Figure 4.25; Man-Whitney, $p$<0.05, Table 4.17). However, both the natural and 2003 gravel treatment sites were characterised by a high mean percentage of fine grained sediment ($D$<1 mm), frequently in excess of 14%. Moreover, few samples from individual gravel sites had low mean percentage fine sediment ($D$<1 mm) compositions between 5 and 20 cm depth. Those that did came from sites 2009A and 2009J. The percentage sediment <1 mm of individual sites within rehabilitation gravel treatments were not distinct, unlike those of natural gravels which displayed significant diversity (Kruskal-Wallis, $p$<0.05, Table 4.17). Elevated fine sediment (<1 mm) composition of the Whey Curd site increased the overall average for natural treatment gravel (Figure 4.9). The 2003 gravel treatment was composed of a significantly greater proportion of <1 mm sediment within the incubation substrate than the 2009 treatment sediments; 18.4% and 9.8% respectively (Man-Whitney, $p$<0.05, Table 4.17).

### 4.6.2 Sand index: a quantitative indicator of spawning habitat quality

As a quantitative spawning gravel quality index, the sand index (SI) provides a measure of the contribution of coarse sand (0.5≤$D$<2 mm) to fine sand ($D$<0.5 mm) in spawning substrates; the lower the calculated value the greater the potential for alevin emergence (Peterson and Metcalfe, 1981). SI scores between 5-20 cm depth across all sites indicated poor to mediocre spawning health (Table 4.18). Surface sediments consisted of mostly gravel within the size range 64>$D$≥16 mm and as such SI scores indicated excellent spawning habitat. However the SI values indicated a marked deterioration of spawning quality with increased depth, particularly within cores sampled from natural gravel sites. Based on the relative contribution of sand, the 2009 treatment sites provided relatively good surface substrate for alevin to emerge from.

**Table 4.17** Summary results of Kruskal-Wallis and Mann-Whitney U analysis for percentage composition difference of fine sediment ($D$>1 mm) within embryo incubation substrate (5-20 cm) between gravel treatments, sites (s) and cores (c). *1* indicates a significant or positive test result, *0* indicates a negative result and - indicates no test. Where Kruskal-Wallis analysis indicated a negative result, no pairwise Mann-Whitney U tests were performed.

| | **Kruskal-Wallis** | **Mann-Whitney U** | |
|---|---|---|---|
| | | **2003** | **2009** |
| **Treatment** | 1 | - | - |
| **Natural (s)** | 1 | 1 | 1 |
| **Fort (c)** | - | - | - |
| **Water Hall (c)** | - | - | - |
| **Whey Curd (c)** | - | - | - |
| **2003 (s)** | 0 | - | 1 |
| **2003A (c)** | - | - | - |
| **2003B (c)** | - | - | - |
| **2003C (c)** | - | - | - |
| **2009 (s)** | 0 | - | - |
| **2009A (c)** | - | - | - |
| **2009D (c)** | - | - | - |
| **2009F (c)** | - | - | - |
| **2009J (c)** | - | - | - |

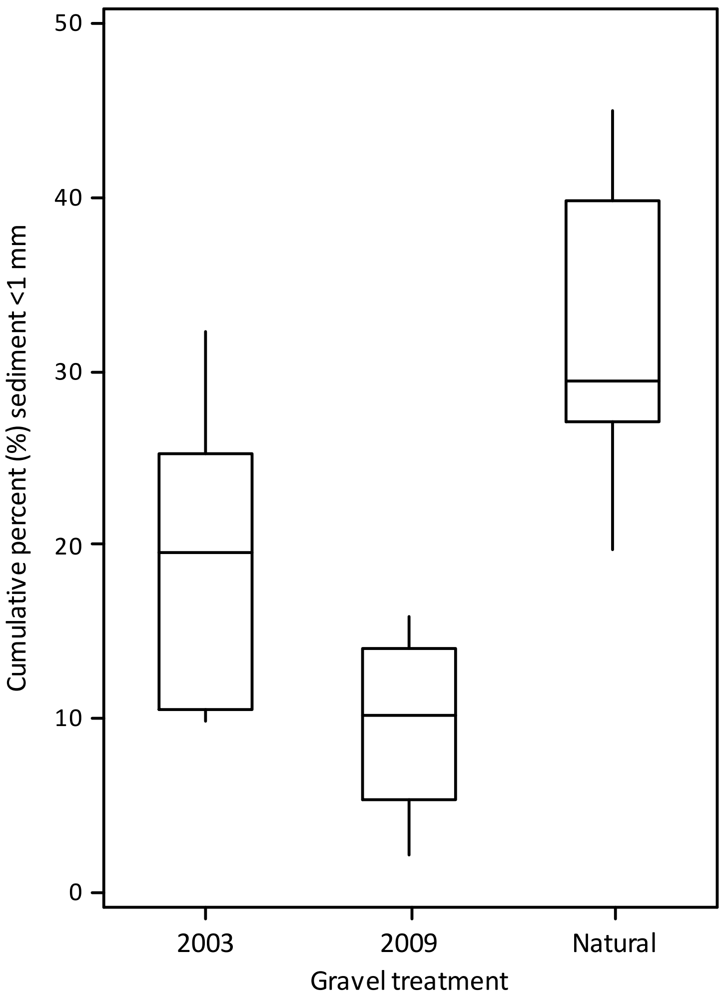



**Figure 4.25** Boxplot of cumulative percentage sediment <1 mm within the embryo incubation zone, 5-20 cm substrate depth, for each gravel treatment. Outliers have been removed from the plot to expand the y-scale of the main data.

**Table 4.18** Summary of the sand index (SI) for individual gravel freeze cores for each 5 cm increment, including core mean ± standard deviation. SI is a quantitative spawning gravel quality index based on the relative contribution of coarse sand (0.5≤*D*<2 mm) to fine sand (*D*<0.5 mm). There was no sample at 20-30 cm for core 2003B due to a core blockage.

| Site | | Depth (cm) | | | | | | Mean ± SD | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | 0-5 | 5-10 | 10-15 | 15-20 | 20-25 | 25-30 | | | |
| Fort | 1 | 0.71 | 2.67 | 4.13 | 3.79 | 3.81 | 4.65 | 3.29 | ± | 1.42 |
| | 2 | 0.58 | 2.95 | 3.98 | 4.53 | 3.33 | 4.23 | 3.27 | ± | 1.44 |
| | 3 | 3.40 | 6.12 | 5.66 | 4.91 | 4.77 | 5.93 | 5.13 | ± | 1.01 |
| W.Curd | 1 | 2.28 | 0.83 | 3.33 | 5.27 | 8.71 | 9.53 | 4.99 | ± | 3.52 |
| | 2 | 0.48 | 1.65 | 3.71 | 4.43 | 7.13 | 8.22 | 4.27 | ± | 3.01 |
| | 3 | 0.29 | 1.87 | 4.26 | 8.19 | 8.22 | 8.28 | 5.18 | ± | 3.57 |
| W.Hall | 1 | 1.80 | 2.75 | 3.22 | 2.75 | 3.57 | 4.53 | 3.10 | ± | 0.92 |
| | 2 | 1.25 | 2.85 | 3.21 | 3.16 | 2.53 | 3.21 | 2.70 | ± | 0.76 |
| | 3 | 0.17 | 1.77 | 2.51 | 3.70 | 2.65 | 2.52 | 2.22 | ± | 1.18 |
| 2003A | 1 | 0.02 | 0.73 | 0.95 | 2.46 | 0.98 | 2.73 | 1.31 | ± | 1.06 |
| | 2 | 0.04 | 0.22 | 1.21 | 2.11 | 1.79 | 1.58 | 1.16 | ± | 0.85 |
| | 3 | 0.30 | 2.46 | 2.70 | 2.95 | 1.09 | 2.12 | 1.94 | ± | 1.03 |
| 2003B | 1 | 0.27 | 1.55 | 2.60 | 2.45 | 4.81 | 5.72 | 2.90 | ± | 2.03 |
| | 2 | 0.29 | 0.67 | 1.25 | 1.13 | - | - | 0.83 | ± | 0.44 |
| | 3 | 0.05 | 0.44 | 1.20 | 1.77 | 0.89 | 4.76 | 1.52 | ± | 1.70 |
| 2003C | 1 | 1.59 | 2.03 | 1.95 | 2.27 | 2.53 | 4.91 | 2.55 | ± | 1.20 |
| | 2 | 0.49 | 2.77 | 2.12 | 3.53 | 6.10 | 4.18 | 3.20 | ± | 1.91 |
| | 3 | 0.62 | 1.87 | 2.97 | 5.03 | 6.75 | 7.12 | 4.06 | ± | 2.66 |
| 2009A | 1 | 0.02 | 0.04 | 0.17 | 0.69 | 1.39 | 1.88 | 0.70 | ± | 0.78 |
| | 2 | 0.22 | 0.36 | 0.46 | 2.08 | 0.94 | 1.78 | 0.97 | ± | 0.79 |
| | 3 | 0.07 | 0.78 | 1.40 | 1.18 | 0.77 | 0.97 | 0.86 | ± | 0.46 |
| 2009D | 1 | 0.03 | 0.15 | 0.66 | 2.20 | 1.79 | 4.42 | 1.54 | ± | 1.66 |
| | 2 | 0.15 | 1.35 | 1.50 | 1.97 | 1.53 | 1.08 | 1.26 | ± | 0.62 |
| | 3 | 0.10 | 0.87 | 1.85 | 2.27 | 5.73 | 8.54 | 3.23 | ± | 3.24 |
| 2009F | 1 | 0.60 | 1.58 | 1.15 | 1.47 | 1.63 | 1.42 | 1.31 | ± | 0.38 |
| | 2 | 0.24 | 1.30 | 1.34 | 1.33 | 1.02 | 0.83 | 1.01 | ± | 0.43 |
| | 3 | 0.13 | 1.58 | 1.17 | 1.51 | 1.09 | 0.37 | 0.98 | ± | 0.60 |
| 2009J | 1 | 0.62 | 0.71 | 0.65 | 0.68 | 0.56 | 0.58 | 0.63 | ± | 0.06 |
| | 2 | 0.17 | 0.18 | 0.36 | 0.89 | 0.81 | 0.92 | 0.55 | ± | 0.36 |
| | 3 | 0.16 | 0.96 | 0.47 | 1.02 | 1.14 | 2.33 | 1.01 | ± | 0.74 |

## 4.7 Effects of gravel rehabilitation on spawning gravel abundance

The quantitative gravel walkover survey was undertaken to determine naturally available spawning grain-sizes, and to investigate how the introduction of rehabilitation gravel*s* have impacted spawning gravel abundance for migratory and non-migratory *S. trutta*. This continuous survey characterised streambed surface (0-5 cm) sediment over the entire study area (see Figure 2.1, Chapter 2). Grading of discrete gravel sizes between 5-100 mm included the size range of gravels required for both migratory and non-migratory *S. trutta* spawning. The abundance of gravel ($64>D\geq16$ mm) suitable for migratory *S. trutta* was significantly increased through the introduction of rehabilitation gravel*s*, mostly in the lower size range of 40-20 mm (Figure 4.26; Wilcoxon Signed Rank, $p<0.05$, Table 4.19). However, rehabilitation gravel introductions in 2003 and 2009 did not increase the availability of gravel suitable for non-migratory *S. trutta* spawning ($30>D_{50}\geq16$ mm) (Figure 4.26). The surface sediment grain-size was greater in the 2003 rehabilitation gravel than in those installed in 2009, likely due to erosion (removal) of finer grained sediment not retained in surface armouring (Figure 4.27). Natural gravels contained a greater abundance of the smaller sized gravel ($D$ = 10 mm) (Figure 4.26). The most abundant sediment grain-size within the 2003 and 2009 treatments was 30 mm (39.7%) and 20 mm (46.8%) respectively. Both the 2009 and 2003 treatments had relatively similar surface grain-size abundances of gravel within the range $D\geq40$ mm (Figure 4.26).

**Table 4.19** Wilcoxon Signed Rank test summary results of difference in surface gravel sizes suitable for migratory ($60\geq D\geq15$ mm) and non-migratory ($30>D_{50}\geq15$ mm) *S. trutta* spawning before and after rehabilitation work. Results were obtained from the continuous streambed gravel survey.

| Gravel size | n | W | Median | *p*-value |
|---|---|---|---|---|
| $60\geq D\geq15$ mm | 8 | 36 | 373.5 | 0.014 |
| $30>D_{50}\geq15$ mm | 3 | 6 | 707.8 | 0.181 |

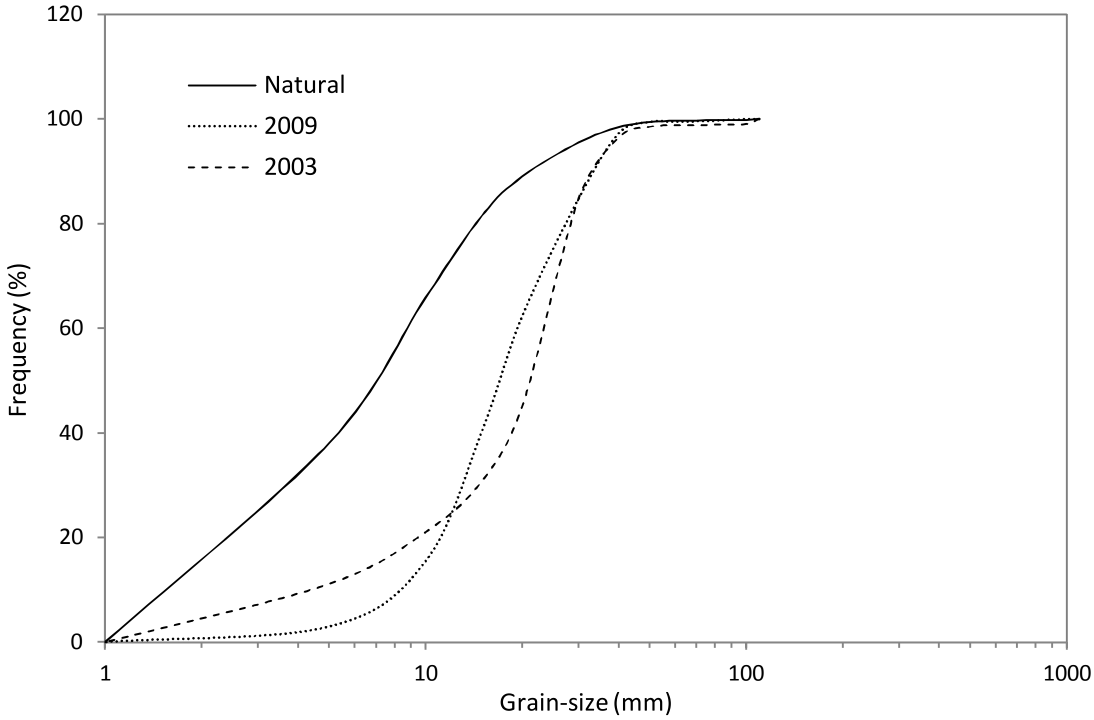
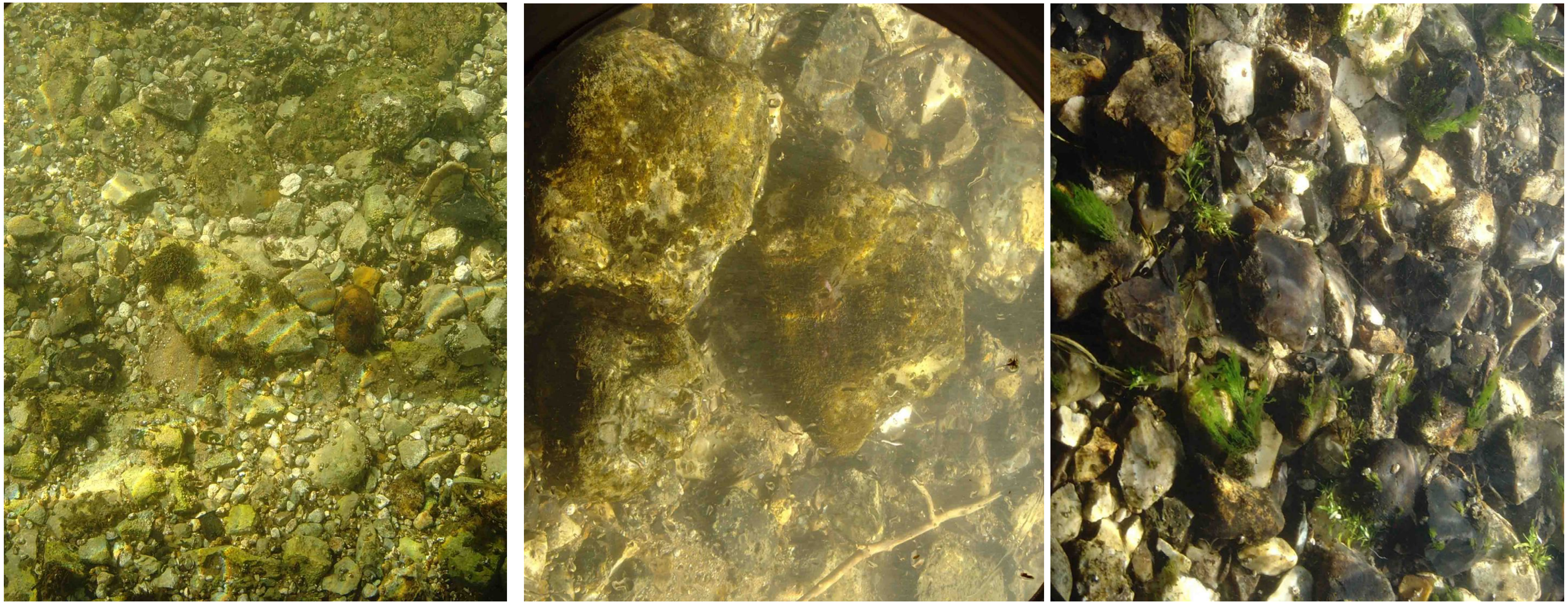



**Figure 4.26** Cumulative grain-size plot of the gravel survey. Note that grain-size frequency (gravel counts) was used. Rehabilitation gravel had a greater surface composition of framework gravels suitable for migratory *S. trutta* spawning. Natural gravels contained finer material.



(a) (b) (c)

**Figure 4.27** Photograph of surface substrate from each gravel treatment: natural gravel at the Water Hall site (a), site 2003C (b), and 2009J (c). Natural gravel treatment was typically poorly sorted, whilst rehabilitation gravel had a greater degree of sorting. Surface sediments in the 2003 rehabilitation gravel were characterised by larger clast sizes than those from 2009 rehabilitation gravel. The smaller sized gravels with the size range 64>*D*≥16 mm, observed in surface sediments in the 2009 rehabilitation gravel site (c), were absent in the 2003 rehabilitation gravel site (b). It is likely that these gravels had been eroded and transported downstream.

## 4.8 Discussion

### 4.8.1 Sediment composition and structure of rehabilitation gravels

Natural and rehabilitation gravel have distinctive sediment compositions. Natural spawning gravels in the River Stiffkey displayed characteristic chalk stream compositional features; spatially variable throughout, a declining sorting co-efficient with increased sediment depth and comparable grain-size $D_{50}$ values (Carling and Reader, 1982; Acornley and Sear, 1999). The degree of sorting observed was dependent on the size of the sediment (Pettijohn, 1957). The grain-size distributions of natural spawning substrate within the River Stiffkey were very poorly sorted and a broad range of sediment sizes was observed. These features were characteristic of poor transport competence likely due to channel modification and over abstraction (Milan and Petts, 1998), deposition of sediment close to the source of input and the influence of storm flow events (Pettijohn, 1957).

Increased sediment depth was associated with smaller grain-sizes for all treatments. However, this increase in smaller grain-sizes with depth was more pronounced, and occurred at shallower depths, within natural substrate. A similar sediment structure has been described for other chalk stream gravel deposits in southern and eastern England (Acornley and Sear, 1999; Milan et al., 2000). Natural spawning gravels typically consisted of smaller grain-sizes than found in rehabilitation gravel, with most gravel restricted to the upper 10 cm of the deposit, forming a well developed surface armour. An elevated fine grained sediment ($D$<1 mm) composition is common within chalk streams substrata, frequently exceeding values observed in upland streams (Carling and Reader, 1982). Suspended and deposited sediment in chalk streams are typically fine grained with dominant sizes in the silt and clay range (Bickerton et al., 1993; Wood and Armitage, 1997; Walling and Amos, 1999). Silt and clay contributions to the sediments composition were considerably greater in two of the three natural spawning gravel treatment sites. A high sand composition of natural spawning gravels observed at the Water Hall site is derived from the readily erodible superficial geology of North Norfolk catchments (Hiscock, 1993; Milan et al., 2000).

Rehabilitation gravel in the River Stiffkey, however, had a sediment composition more closely associated with the hydraulically controlled steeper catchment morphologies, high sediment scour and transport rates of upland gravel-bedded streams (Carling and Reader, 1982). Upland gravel-bedded streams have coarser grained substrate than gravel deposits typically associated with chalk streams (Milan et al., 2000). Similarly, River Stiffkey rehabilitation gravels are characterised by a narrow range of coarse grain-sizes (64>$D$≥16 mm) consisting predominantly of vertically well distributed framework gravels with little stratification and a low fine sediment composition. Upland stream systems do however have a greater hydraulic control that limits excessive accumulation of finer grained sediment within framework gravels (Carling and Reader, 1982). Accumulations of fine grained sediment (<1 mm) observed in the 2003 rehabilitation gravels are symptomatic of the disequilibrium between a readily available and large sediment supply (see below) and a hydraulic regime characterised by low stream power. Fine sediment ($D$<1 mm) accumulations have lowered the sediment sorting co-efficient of the 2003 rehabilitation gravels relative to those installed in 2009.

#### 4.8.2 Rehabilitation gravels and the impact of catchment control mechanisms on spawning habitat suitability

Fine sediment in gravel bedded rivers accumulates due to a combination of factors: a lack of flows with sufficient energy to penetrate deep lying substrata, seasonally high deposition rates, and a stable framework gravel deposit (Acornley and Sear, 1999). Sediment accrual in the framework gravels of all gravel treatment types in the River Stiffkey occurred in a progressive manner through time, although such accumulation of sediment does not necessarily occur in a linear manner (Cefas, 1999). The rate and depth of this accumulation is determined by flow frequency and magnitude as well as the relationship between the size and shape of particles in the associated sediment load and the interstitial void size (Frostick et al., 1984; Knighton, 1984; Lisle, 1989; Charlton, 2008). In turn the interstitial void size is determined by the composition of sediment shape and size (Frostick et al., 1984). The void size of poorly sorted sediment is variable due to the wide range of grain-sizes and the structure of the framework gravel (Lachance and Dubé, 2004). The accumulation of fine grained sediment in framework gravel occurs in a graded manner with larger sizes remaining within the upper substrate and finer grained sediments accumulate deeper (Pettijohn, 1957; Knighton, 1984; Lisle, 1989).

The polymodal nature of the grain-size distribution observed in rehabilitation gravel in the River Stiffkey (Figure 4.19) affords a large difference between framework and matrix grain-sizes enabling fine sediment to infiltrate clean gravel, assuming high porosity, at low velocities and settle deep within the deposit (Lisle, 1989) filling gravel interstices from the bottom up (Frostick et al., 1984). The grain-size distribution observed in spawning sediments is therefore a reflection of the prevalent hydraulic conditions as well as the nature of sediment supply (Reid et al., 1997).

#### 4.8.2.1 Catchment controls: sediment supply and hydraulic regime

The sedimentary character of the River Stiffkey is controlled by processes that occur at the catchment scale (Chapter 3). Typical of chalk streams, the River Stiffkey is susceptible to catchment-derived sediment inputs (Wood and Armitage, 1997; Walling and Amos, 1999; Mainstone et al., 1999; Heywood and Walling, 2003; Walling et al., 2006). The river is sediment supply controlled; the ability of the stream to receive and store excessive catchment-derived sediment loads is high due to the characteristic low stream competence. Erosion and run-off from arable fields, farm tracks and roads, as well as river bank poaching by livestock are all identified sources of sediment in the River Stiffkey. Rain storm events erode and transport vast quantities of sediment from the landscape into the river channel.

Hydraulic controls on sedimentation remain a vital hydrogeomorphic process. The River Stiffkey, like other chalk streams, is characterised by low velocities. Due to the gentle nature of the channel gradient and historic stream management (largely flood mitigation, see Chapter 3), the river has low stream energy (poor transport competence) and therefore a limited ability to erode and transport excessive sediment inputs, as observed in Leopold et al. (1964). Gravel recruitment is therefore limited and a high in-stream fine sediment supply is maintained. Transportation of fine grained sediment can occur at low velocities and as such accumulation in spawning gravel is a function of sediment supply, whilst coarser material transport is capacity-limited and downstream displacement is dependent on elevated velocities (Carling, 1983; Knighton, 1984). The spatial variability of the sedimentation process and gravel composition in the River Stiffkey is a function of the temporal nature of the hydraulic regime.

In rivers similar to the River Stiffkey, finer grained sediments were transported and deposited during summer base flows with coarser sediments mobilised during higher winter stream velocities (Frostick et al., 1984; Acornley and Sear, 1999). There is a net accumulation of fine grained sediments in chalk streams during the winter months associated with the greater sediment run-off (Walling and Amos, 1999). Greater rates of deposition into gravel beds occur when discharge and sediment transport rates are high (Carling, 1983; Frostick et al., 1984; Acornley and Sear, 1999). Summer base flows slowly redistribute catchment-derived sediments downstream (Carling and Reader, 1982; Walling and Amos, 1999). During the summer months a physically comparable stretch of the River Test in Hampshire had low sediment transport rates, which peaked during winter when >95% of the annual suspended sediment load was mobilised (Acornley and Sear, 1999). A greater mean deposition rate occurred during the winter months, 0.5-1.0 kg $m^{-2}$ $day^{-1}$, compared to the mean summer rate of 0.02 kg $m^{-2}$ $day^{-1}$ (Acornley and Sear, 1999). High magnitude rainfall events erode and transport fine sediments stored within the stream channel creating temporal and spatial variability, frequently referred to as the 'pulsing' of sediment through stream reaches during summer months (Frostick et al., 1984; Walling and Amos, 1999). Walling and Amos (1999) argued that transport and deposition processes are not as important in non-calcareous catchments where sediment pulses through the stream and little is stored between high discharge events. Rehabilitation gravel in the River Stiffkey effectively reduced localised depth but failed to sufficiently increase water velocity and thus increased susceptibility to sediment deposition. These sites will therefore not be able to sustain their physical integrity. In this respect the sedimentary character of the rehabilitation gravels are sediment-supply controlled and defined by those processes operating at the catchment scale. In association with the hydraulic regime, excessive catchment-derived sediment loadings have physically altered rehabilitation gravel in the River Stiffkey with implications for the suitability and availability of *S. trutta* spawning habitat.

#### 4.8.2.2 Suitability and availability of rehabilitation gravels for non-migratory and migratory *S. trutta* spawning

The spatially variable sediment composition of the natural and 2009 rehabilitation gravels likely creates a high quality habitat, as observed in Pasternack et al. (2004). However, accrual of fine grained sediment has lowered the spatial variability of the 2003 treatment. Abundance of fine sediment ($D$<1 mm) in the spawning matrix is a major factor in the decline of native salmonid stocks, affecting productivity at the egg and juvenile stage (Turnpenny and Williams, 1980; Mann et al., 1989; Crisp, 1993; Acornley and Sear, 1999; Hendry et al., 2003). A high composition of fine sediment reduces interstitial voids, and thereby intragravel permeability, inhibiting embryo development through reduced capacity of water to deliver dissolved oxygen and remove associated metabolic wastes (Theurer et al., 1998; Greig et al., 2005a; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006). Furthermore, reduced $D_{50}$ grain-sizes are associated with impacts on ecological community structure; loss of (invertebrate) species diversity and abundance (Shaw and Richardson, 2001).

The deposition of fine sediment into gravel deposits in the River Stiffkey, like many other chalk streams, exceeds the 14% threshold of a healthy *S. trutta* spawning habitat. Natural and rehabilitation gravel within the River Stiffkey are characterised as having a high mean abundance of sediments *D*<1 mm and poor sand indices within the embryo development zone (5-10 cm), particularly so for the natural and 2003 treatments. Just two 2009 rehabilitation gravel sites, 2009A and 2009J, had a low (<6%) mean fine sediment (*D*<1 mm) contribution. However, given the high deposition of fine sediment observed in the River Stiffkey, it is expected that all sites in the 2009 treatment will in the short term accumulate greater quantities of fine grained sediment throughout the vertical extent.

Rates of *S. trutta* growth are associated with the availability of food resources (Elliot and Hurley, 2000). Chalk streams are highly productive ecosystems and as such *S. trutta* are faster growing than those within non-calcareous river systems (Mann et al., 1989; Wootton, 1998). Non-migratory *S. trutta* attain sexually maturity at a smaller body size than their migratory morphs (Klemetsen et al., 2003). In their study on small streams Jonsson et al. (2001) found mean body length at sexual maturation ranged between 160-240 mm and an age of 2+ to 4+. There was a good association between the required spawning gravel $D_{50}$ (20 mm) based on the weighted average length of sexually mature *S. trutta* in the River Stiffkey, and the observed natural gravel $D_{50}$ grain-size. However, it is likely that the majority of this is retained in surface armouring. The 2003 rehabilitation gravels have lost the surface layer (5 cm) of smaller more suitably sized gravel ($30>D_{50}\geq16$ mm) for non-migratory *S. trutta*. Moreover, this rehabilitation gravel treatment had a surface grain-size $D_{50}$>20 mm, but between 15-25 cm depth the $D_{50}$ approximates 20 mm. Although non-migratory *S. trutta* spawn in a shallow gravel layer due to their small size at maturity, depths exceeding 10-15 cm are too great for successful reproduction (Milan et al., 2000; Armstrong et al., 2003). The 2003 rehabilitation gravels therefore are no longer suitable for spawning by non-migratory *S. trutta*.

Like other chalk streams (see Mann et al., 1989), natural gravels in the River Stiffkey are limited; previous flood mitigation management has removed considerable gravel bed habitat, and excessive inputs of fine sediment (*D*<1 mm) have smothered much of the remaining gravel habitat. The characteristic lack of gravel and abundance of sediments *D*<1 mm have skewed the natural grain-size distributions towards a smaller $D_{50}$. Distribution cumulative percentile statistics indicated a scarcity of suitably sized spawning gravel for non-migratory *S. trutta* in the River Stiffkey that extend deep enough into the substrate for widespread and consistently successful embryo development (Tables 4.8-4.10). However, additions of approximately 800 tonnes of rehabilitation gravel to the lower reaches of the river have significantly increased the availability of gravel (64>$D$≥16 mm) forcing an increase in $D_{50}$ with a greater proportion of gravel extending further into the substrate than observed in naturally occurring gravel deposits. Although gravel rehabilitation has significantly increased the availability of spawning substrate for migratory *S. trutta*, increases in gravel 30>$D_{50}$≥16 mm, however, did not increase the total abundance of suitably sized gravel for non-migratory *S. trutta*. The potential for recruitment to the migratory *S. trutta* population from an augmented non-migratory *S. trutta* population (increased pressure and competition for resources) has therefore not been improved.

### 4.8.3 Rehabilitation gravel succession

Inconsistency and the ephemeral nature of rehabilitation gravel are perhaps the greatest barriers to rehabilitation gravel projects (Merz et al., 2004; Barlaup et al., 2008; Pedersen et al., 2009). Poorly sorted streambed sediments are frequently packed into a tight structure creating greater stability that requires greater force to mobilise than a well sorted grain-size distribution (Reid et al., 1997). Pasternack et al. (2004) argued that a more heterogeneous mix of sediment sizes should therefore be included in rehabilitation gravel to provide greater stability, prevent erosion of the smaller mobile gravel sizes and inhibit or reduce finer grained sediment accrual. However, gravel bed stability is controlled by hydraulic processes and the nature and supply of sediment within the catchment (Werritty, 1997). In this manner catchment control mechanisms determine the morphosedimentary nature of rehabilitation gravel in the River Stiffkey.

During the summer months stream velocity in the River Stiffkey is maintained at a low base flow velocity that underpins the establishment of well developed surface armouring of the streambed. Surface armouring can prevent ingress of fine grained sediment and in some instances creates unfilled voids beneath surface substrate (Frostick et al., 1984). The River Stiffkey rehabilitation gravel*s* have well sorted sediments and therefore less stability associated with an increased susceptibility to fine grained sediment accretion. Redistribution of smaller more mobile gravels (30>$D_{50}$≥16 mm) was assumed in the surface substrate of rehabilitation gravel*s* that lack the stabilising effect of surface armour. Clasts ≥64 mm, used to anchor the gravel structures in place (Figure 4.1), were significantly closer to surface substrate in the 2003 than in the 2009 rehabilitation gravel sites due to erosion of surface sediments.

Although these clasts will significantly increase bed stability (Milan et al., 2000), the 2003 rehabilitation gravels have lost the overlaying gravel through erosion and downstream transport, and as such are a less suitable spawning habitat. Erosion of the smaller more mobile gravels suitable for non-migratory *S. trutta* spawning ($30>D_{50}\geq16$ mm) was however evident for both the 2009 and 2003 rehabilitation treatments. Surface sediment armouring has prevented this from happening in the natural gravel treatment.

Chalk stream gravel deposits that are artificially 'cleaned' of fine sediment will regress back to a high fine sediment (<1 mm) composition as a function of time (Acornley and Sear, 1999; Cefas, 1999), sediment supply and hydrological regime. Acornley and Sear (1999) established that it took 25 days for cleaned gravel to revert back to pre-cleaned conditions in the River Test, a chalk stream in Southern England. The 2003 rehabilitation gravels have accrued greater volumes of fine ($D$<1 mm) sediment than those installed in 2009 due to a longer period of exposure to stream sedimentation processes. The quantity of fine sediment is comparable to those observed in natural treatment sites. Prolonged accumulation of fine sediment has altered the distribution $D_{50}$ of the 2003 rehabilitation gravels relative to the 2009 rehabilitation gravels; the decrease in $D_{50}$ with depth is more pronounced in the 2003 treatment substrate. Given the high loadings of catchment-derived sediment in the river channel, it can be deducted that rehabilitation gravel in the River Stiffkey undergo a succession from a very well sorted gravel type, similar to deposits associated with upland streams with a narrow range of coarse gravel, towards a poorly sorted deposit composed of a broader range of sediment sizes.

The initial compositional variability, observed within the 2009 treatment, has been dampened by vertical and horizontal accrual of fine grained sediment (<1 mm) as observed within the 2003 treatment. Rehabilitation gravel installed into the River Stiffkey were graded similarly to select suitable spawning gravel sizes prior to installation ($40\leq D\leq10$ mm), and constructed to similar specifications (T. Jacklin, pers. comm., 17/01/2011). Greater fine grained sediment deposits in gravels installed in 2003 provided a good indication of the longevity and suitability of rehabilitation gravel as a viable spawning habitat. At the time of sampling (2011), the sedimentary character of the 2009 rehabilitation gravels provided an appreciably more suitable spawning habitat than those installed in 2003. As such rehabilitation gravel is likely suitable for salmonid spawning in the short-term only. The 2009 rehabilitation gravels are at risk of accumulating an abundance of catchment-derived fine grained sediment and deteriorating into a poor spawning habitat. It is estimated that <10 years after installation these gravels will be unsuitable for viable *S. trutta* spawning.

A similar rehabilitation gravel morphosedimentary succession, with an associated decline in *S. trutta* spawning suitability, was observed in the Moosach River, southern Germany (Pulg et al., 2013). The Moosach River drains a Quaternary limestone gravel catchment and sediment loading is high due to historic modification, impoundments and land-use. Sedimentary conditions of rehabilitation gravel (16>*D*>32 mm) installed in pool-riffle structures were monitored between 2004-2008. Very favourable sedimentary conditions were maintained for 2 years post installation, but accrual of fine sediment (size not specified) degraded the spawning condition rapidly after 4 years with a completely unsuitable spawning environment estimated between 5-6 years post gravel installation (Pulg et al., 2013). Introductions (10-200 cm deep) of rehabilitation gravel (25>*D*>125 mm) in an alternate bar-like fashion to the heavily impacted Mokelumne River (damming, gold mining and agricultural land-use) in California, USA, significantly increased elevation, velocity and intragravel permeability 2 years post introduction (Merz and Setka, 2004). However, sediment (≤8 mm) accrual >2 years post introduction increased to levels similar to natural gravels. Similar fine grained sediment accumulations within rehabilitation gravel in the River Stiffkey inhibit the long-term quality of these installations as a suitable spawning habitat.

## 4.9 Conclusion

River Stiffkey hydrogeomorphology is sediment supply dominated due to poor hydraulic controls and an abundant supply of catchment-derived sediment. The morphosedimentary character of rehabilitation gravel in the River Stiffkey reflects the nature of catchment processes. Rehabilitation gravel undergoes a sediment-driven succession from a spatially variable gravel-rich composition to a non-spatially variable state with a greater proportion of fine sediment distributed vertically and horizontally throughout the deposit. The installation of rehabilitation gravels failed to significantly increase stream velocities. Flushing of fine sediment from gravel treatments is limited due to low summer velocities and an abundance of fine sediment eroded from the catchment and deposited in the channel during winter. The suitability of 2003 rehabilitation gravels for *S. trutta* spawning is currently poor. Given that rehabilitation gravel have good bed stability due to the presence of clasts ≥64 mm, the poor spawning quality will continue to exist in this state until very high, but rare, stream velocity erodes out deep laying fine sediment, or further management intervention scours the deposit. However, any alternate physical state will not be stable and will regress back given the hydraulic control and sediment supply of the river channel. Holistic sediment management at the catchment scale will however improve the sediment quality of the rehabilitant gravel for salmonid spawning.

Given similar conditions the spawning value of the 2009 rehabilitation gravels will devalue too over the short-term (<10 years) as their physical integrity is altered in the same way as the 2003 rehabilitation gravels. As such rehabilitation gravel fails to provide a high quality self-regulating spawning habitat for *S. trutta*. Given that a suitable gravel size range ($30>D_{50}\geq16$ mm) currently exists within natural gravels in the river, recruitment to the migratory *S. trutta* population through augmentation of the non-migratory population could have been achieved with cheaper, long-term and more sustainable approaches, such as rehabilitation of existing natural spawning habitat. Indeed management strategies aimed at decreasing catchment run-off and increasing hydraulic controls would have been more suited to this purpose.

# 5 Quantification of *S. trutta* embryo survival in rehabilitation gravels: effects of catchment control variables

## 5.1 Introduction

Catchment and hydrogeomorphological processes have direct impacts on *Salmo trutta* ecology, especially in relation to reproductive stages of its life cycle. *S. trutta* embryo development requires complex chemical and biological interactions that are controlled by spawning sediment composition. Alteration of sediment composition affects these interactions with consequent impacts on embryo development (Rubin, 1992; Kondolf, 2000; Armstrong et al., 2003; Ojanguren and Braña, 2003; Greig et al., 2005a).

In this chapter the physical controls that determine population recruitment at the egg stage of the *S. trutta* life cycle are investigated. This chapter aims to:

- quantify the biological suitability of rehabilitation gravel for *S. trutta* spawning based on embryo survival
- determine the biological response to the morphosedimentary succession observed in rehabilitation gravel installed in the River Stiffkey

The results of an extensive two year egg-box study are presented and considered in relation to sediment composition, providing an indication of spawning quality and functional responses to the morphological succession of rehabilitation gravel observed in Chapter 4. The survival of embryo in rehabilitation gravel was determined by inserting 4 egg-boxes, each containing 50 eyed *S. trutta* eggs, into artificially cut redds and by counting alevin survival after a predetermined time, defined by stream temperatures (see section 2.3.3, Chapter 2). In order to determine the feasibility of an egg-box study design and methods in the River Stiffkey, as well as to verify the required replication of redds and sites, a trial study was initially conducted in 2011 (egg-boxes installed 3 February, recovered 25 March). Discharge during egg-box emplacement and recovery was 0.97 $m^3 s^{-1}$ and 0.63 $m^3 s^{-1}$ respectively (Figure 5.1). Five sites were used during this study: rehabilitation gravel 2003A, 2003C, 2009A, 2009J and the natural gravel site Water Hall (Figure 2.1, Chapter 2). Rehabilitation gravel was installed in modified (straightened and dredged) river reaches. Natural treatment gravels were located in shallower less modified reaches and typically had smaller sediment grain-sizes. A total of 7 redds were artificially cut on each gravel site to ensure good surface area coverage (Figure 2.7, Chapter 2). Redd dimensions adhered to those outlined in Crisp and Carling (1998). Stream velocity at several points around each redd was measured consistent with Crisp and Carling (1998). Similar egg-box studies have used comparatively fewer redds per site. Harris (1973) installed egg-boxes into a total of 8 artificially made redds constructed next to natural redds. Dumas and Marty (2006) constructed 4 redds on each of 3 sites. Similarly, Syrjänen et al. (2008) buried five egg-baskets per site (4 sites in total), whilst Pulg et al. study (2013) emplaced five egg-boxes containing 190 embryo each per site over 5 sites. Turnpenny and Williams (1980) only buried 3 egg-boxes on 7 sites.

Based on the 2011 trial egg-box study, the 2012 study (egg-boxes installed 11 January and recovered 19 March) had an increased site replication and provided a balanced study design; 3 sites from each gravel treatment were used covering nine sites in total. A reduced discharge compared to the 2011 study of 0.30 $m^3 s^{-1}$ was observed during embryo emplacement and 0.36 $m^3 s^{-1}$ whilst recovering egg-boxes. All 3 gravels sites in the 2003 rehabilitation treatment were included (2003A, 2003B, 2003C), sites 2009A, 2009D and 2009J from the 2009 rehabilitation treatment and natural gravels Whey Curd, Water Hall and Fort represented the natural treatment. Each of these sites had 7 redds artificially cut, and four egg-boxes installed into each, with a total of 200 eyed *S. trutta* eggs per redd, as used in the 2011 trial study. Stream velocity was measured at four points around each redd (Figure 2.8, Chapter 2); directly in front of and to the sides of the redd pit, as well as on top of the highest point on the tailspill at 60% depth, consistent with Crisp and Carling (1998). Unlike the 2011 trial study, redd sediment composition was sampled before and after embryo incubation by means of freeze cores. The composition of redd sediment within each treatment was assumed to be similar prior to embryo installation and as such a single redd was cut and sampled from the upstream most site from each gravel treatment directly after embryo installation, 11 January 2012. Redd sediment composition post embryo incubation was also sampled by means of freeze cores; redds were cut at all sites within each treatment and cored immediately after embryo recovery, 19 March 2012. This provided an indication of sediment accumulation during the embryo incubation period. Specific redds were cut for sediment samples and no *S. trutta* eggs were installed in these.

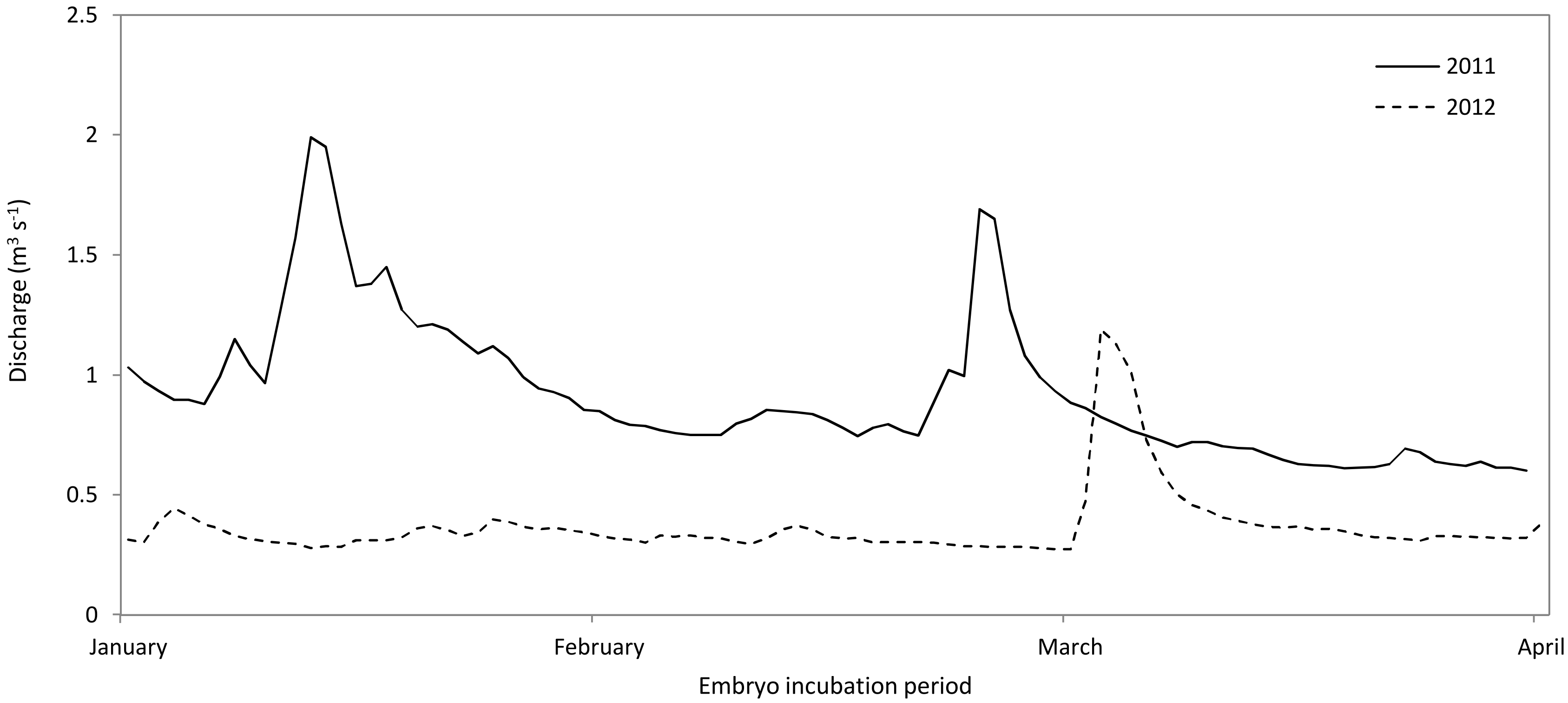



**Figure 5.1** Stream discharge (m $s^{-1}$) in the River Stiffkey during the embryo incubation period, January to March 2011 and 2012. Discharge was measured at the Environment Agency gauging station near Warham. Greater discharge during 2011 resulted in a relatively higher discharge that year. The early months of 2012 had very low rainfall.

## 5.2 2011 trial study: ETF survival and lessons learnt

Contents of egg-boxes were sorted into four categories; dead alevin, live alevin, dead eggs, live eggs. No dead alevin or live eggs were recovered. Most eggs were not recovered as they had either decayed or were likely preyed on by leeches. Live alevin, dead eggs and egg chorion (outermost egg membrane) as well as fungal masses were all recovered (Figure 5.2). Egg-to-fry (ETF) survival per egg-box was low and the frequency of egg-boxes that did not record any survival was high (Figure 5.3). The ETF survival distribution was non-normal and strongly positively skewed.

Mean ETF survival for gravel treatments were 20.5%, 2.7% and 6.1% for the natural, 2003 and 2009 rehabilitation gravel treatments respectively (Table 5.1). Redds cut on the natural gravels in the Water Hall site had significantly higher ETF survival compared to sites 2003A, 2003C and 2009A (Figure 5.4a; Mann-Whitney, $p<0.05$, Table 5.2). ETF survival was significantly low in the 2003 and 2009 rehabilitation treatment compared to the natural gravel treatment overall (Figure 5.4b; Mann-Whitney, $p<0.05$, Table 5.2).

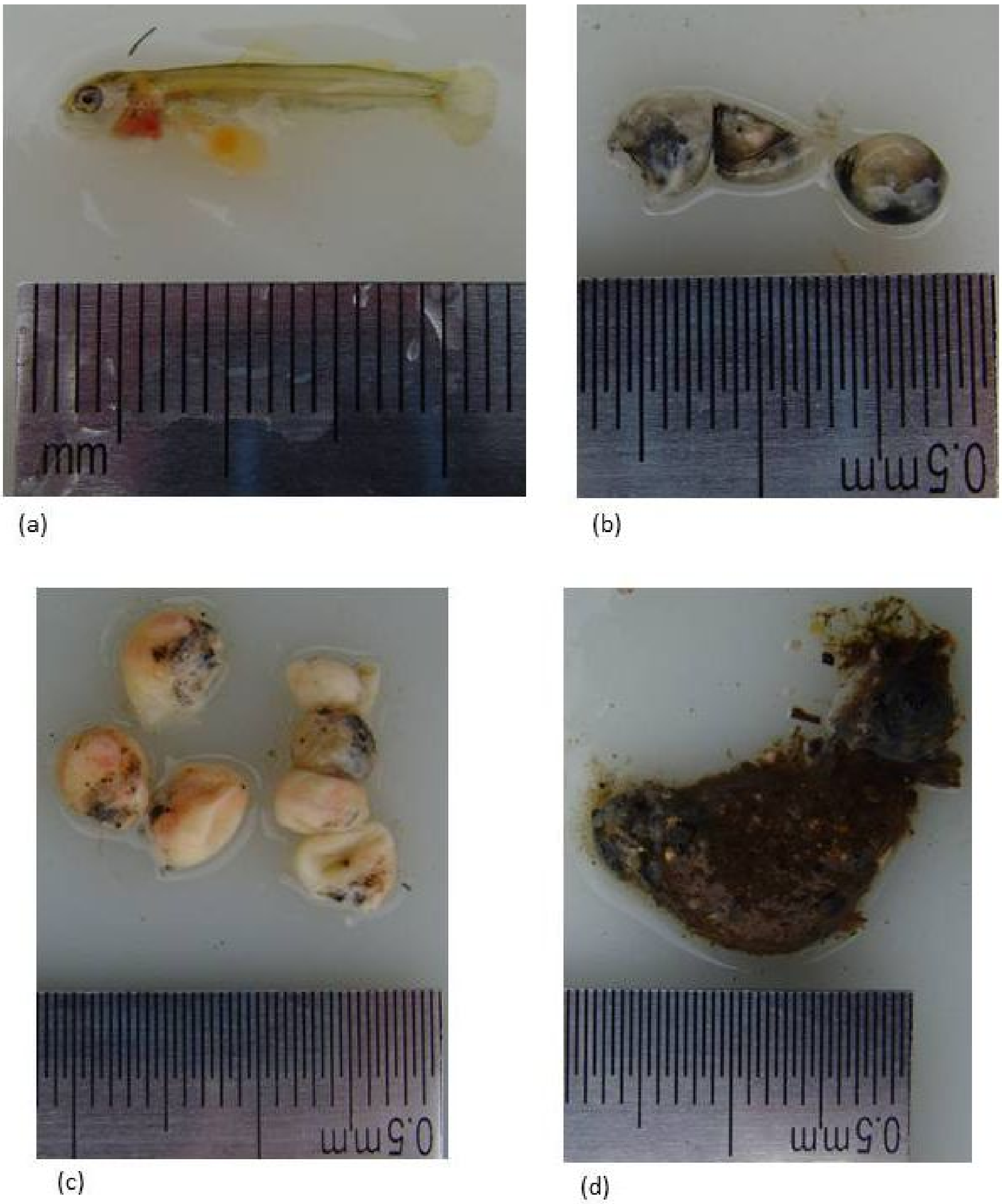



**Figure 5.2** Typical material recovered from the egg-boxes: yolk sac fry (a), empty embryo chorion (b), dead embryos (c) and unidentified fungal matter (d).

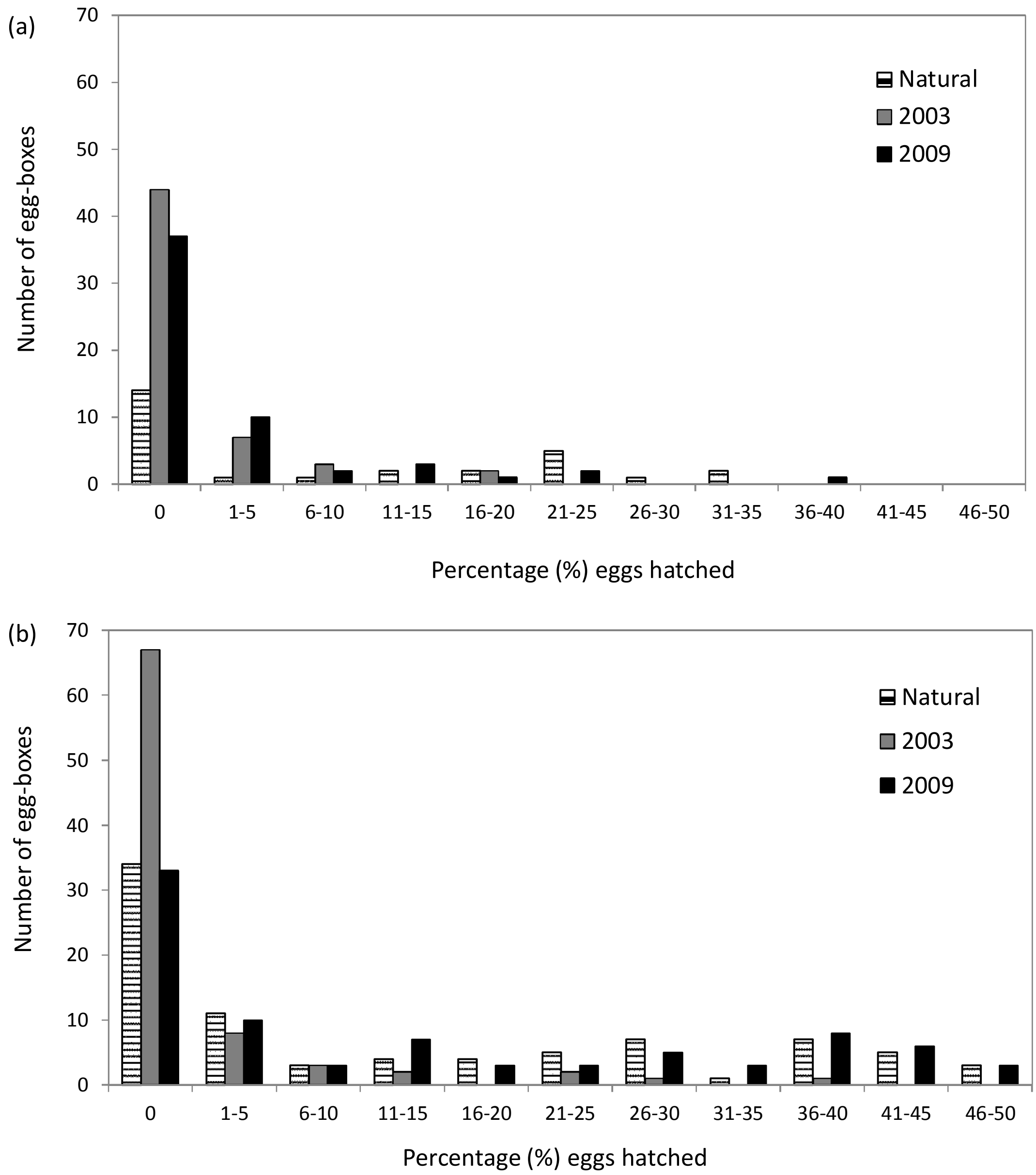



**Figure 5.3** Frequency histogram of egg-to-fry (ETF) survival (%) per egg-box within each treatment in 2011 (a) and 2012 (b). There were high numbers of egg-boxes recovered with 100% mortality (0 survival) in both years.

Total mean ETF survival was poor with just 7.6% of embryos developing into alevins (Table 5.1). However, the 2011 pilot study included only a single natural treatment site (Water Hall) and two sites within each rehabilitation gravel treatment. This biased results when examined at the treatment-level with ETF survival favouring the natural treatment. Based on the 2011 trial study results, a balanced study design that included greater replication at the site level were used for the 2012 study. Further, sediment composition of the redd incubation environment, absent during the 2011 trial study, was sampled in association with ETF survival to strengthen the 2012 egg-box study. Stream velocity at several points around the redd were measured, consistent with Crisp and Carling (1998). During the 2011 trial, redd velocity was only measured post-embryo incubation. Redd velocities were measured twice during the 2012 study, pre- and post-embryo incubation.

### 5.3 *S. trutta* embryo survival in rehabilitation gravel

All egg-boxes were recovered, except the rear left box from redd 2 at site 2009A. As observed in the 2011 trial study, there was a high frequency of egg-boxes with zero (0) ETF survival (Figure 5.3b), and the distribution was similarly non-normal and strongly positively skewed. However, unlike the 2011 trial study, there was greater ETF survival per egg-box in mostly natural and 2009 rehabilitation gravel treatments that increased the positively skewed tail. Overall, low ETF survival was common across sites and treatments (Figure 5.5; Table 5.1). Natural gravels at both the Water Hall and Fort sites had significantly greater ETF survival than observed at the Whey Curd site (Figure 5.5a; Mann-Whitney, $p$<0.05, Table 5.2). Mean ETF survival in gravels on the Whey Curd site were very low, just 4.5% (Table 5.1). The 2003 rehabilitation gravels sites all had low mean ETF survival, 0.8%, 5.4%, 6.2% for 2003A, 2003B and 2003C respectively (Figure 5.5a; Table 5.1). Conversely, sites within the 2009 rehabilitation gravel treatment, 2009A, 2009D and 20009J all had significantly greater ETF survival (Figure 5.5a; Mann-Whitney, $p$<0.05, Table 5.2). However, ETF survival in site 2009A was not significantly different from the 2003 rehabilitation sites (Figure 5.5a). ETF survival in all 4 of these sites was particularly low, with a mean of 6.7% from site 2009A (Table 5.1).

At the treatment level, both the natural and 2009 rehabilitation gravel had significantly greater ETF survival compared to the 2003 rehabilitation gravel treatment (Figure 5.5b; Mann-Whitney, $p$<0.05, Table 5.2). No similar difference in ETF survival was observed between the natural and 2009 rehabilitation treatments. A mean ETF survival for the natural, 2003 and 2009 rehabilitation gravel treatments were 27.0%, 4.1% and 28.8% respectively (Table 5.1). Overall, mean ETF survival improved significantly in 2012 (Mann-Whitney, $p$<0.05, Table 5.2) with 20% compared to a mean of 7.6% during 2011 (Table 5.1).

Only a small number of sites within the study area offer a suitable spawning habitat for *S. trutta* and as such regulate population recruitment. During the 2011 trial, natural gravels at the Water Hall site accounted for 54% of cumulative ETF survival, while both sites within each of the 2003 and 2009 rehabilitation gravel treatments had very low ETF survival; 0.3% and

5.1% for sites 2003A and 2003C, and 5.1% and 7.1% for sites 2009A and 2009J respectively (Figure 5.6a). Less than 50% of gravel sites in the 2012 study were responsible for 87% of the observed cumulative ETF survival; natural gravels at Water Hall and Fort as well as 2009 treatment sites 2009D, 2009J accounted for 24%, 19%, 15% and 29% respectively (Figure 5.6b).

**Table 5.1** Summary ETF survival (%) illustrating mean ± SD and p-values of the Anderson-Darling test of normality for sites and treatments. No sites had high ETF survival only very few had modest ETF survival. Total eggs installed into each site are indicated in the *# eggs* column, whilst the number of eggs that developed into alevin are indicated under the *Survival* column. All sites except 2009J during the 2012 study were non-normally distributed, as indicated by Anderson-Darling tests under the *p-value* column. * A single egg-box was not recovered.

| | | | | Sites | | | | | Treatments | | | | Study | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | Survival (%) | | | | Survival (%) | | | | Survival (%) | | |
| Year | Treatment | Site | # eggs | Survival | Mean | ± | SD | p-value | Mean | ± | SD | p-value | Mean | ± | SD |
| 2011 | Natural | W. Hall | 1400 | 287 | 20.5 | ± | 24.1 | <0.005 | 20.5 | ± | 24.1 | <0.005 | | | |
| | 2003 | 2003 A | 1400 | 4 | 0.3 | ± | 1.2 | <0.005 | | | | | | | |
| | | 2003 C | 1400 | 72 | 5.1 | ± | 10.2 | <0.005 | 2.7 | ± | 7.6 | <0.005 | | | |
| | 2009 | 2009 A | 1400 | 72 | 5.1 | ± | 10.8 | <0.005 | | | | | | | |
| | | 2009 J | 1400 | 99 | 7.1 | ± | 16.8 | <0.005 | 6.1 | ± | 14.0 | <0.005 | 7.6 | ± | 16.1 |
| 2012 | Natural | W. Curd | 1400 | 63 | 4.5 | ± | 15.1 | <0.005 | | | | | | | |
| | | W. Hall | 1400 | 594 | 42.4 | ± | 36.5 | <0.005 | | | | | | | |
| | | Fort | 1400 | 477 | 34.1 | ± | 29.6 | 0.015 | 27.0 | ± | 32.6 | <0.005 | | | |
| | 2003 | 2003 A | 1400 | 11 | 0.8 | ± | 2.5 | <0.005 | | | | | | | |
| | | 2003 B | 1400 | 76 | 5.4 | ± | 16.4 | <0.005 | | | | | | | |
| | | 2003 C | 1400 | 87 | 6.2 | ± | 14.4 | <0.005 | 4.1 | ± | 12.8 | <0.005 | | | |
| | 2009 | 2009 A | *1350 | 90 | 6.7 | ± | 16.8 | <0.005 | | | | | | | |
| | | 2009 D | 1400 | 377 | 26.9 | ± | 33.2 | <0.005 | | | | | | | |
| | | 2009 J | 1400 | 730 | 52.1 | ± | 30.1 | 0.100 | 28.8 | ± | 33.2 | <0.005 | 20.0 | ± | 29.9 |

**Table 5.2** Summary results of Kruskal-Wallis and Mann-Whitney U analysis for the difference of egg-to-fry (ETF) survival between treatments and sites in both 2011 and 2012. 1 indicates a significant or positive test result, 0 indicates a negative result and - indicates no test.

| | | **Kruskal-Wallis** | **Mann-Whitney U** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | **2003** | **2009** | **Fort** | **Water Hall** | **Whey Curd** | **2003B** | **2003C** | **2009A** | **2009D** | **2009J** | **2012** |
| **2011** | **Treatment** | 1 | - | - | - | - | - | - | - | - | - | - | - |
| | **Natural** | - | 1 | 0 | - | - | - | - | - | - | - | - | - |
| | **2003** | - | - | 0 | - | - | - | - | - | - | - | - | - |
| | **2003A** | - | - | - | - | 1 | - | - | 1 | 1 | - | 1 | - |
| | **2003B** | - | - | - | - | 1 | - | - | - | - | - | - | - |
| | **2003C** | - | - | - | - | 1 | - | - | - | 0 | - | - | - |
| | **2009A** | - | - | - | - | 1 | - | - | - | - | - | - | - |
| | **2009J** | - | - | - | - | 0 | - | - | 0 | 0 | - | - | - |
| **2012** | **Treatment** | 1 | - | - | - | - | - | - | - | - | - | - | - |
| | **Natural** | - | 1 | 0 | - | - | - | - | - | - | - | - | - |
| | **Fort** | - | - | - | - | 0 | - | - | - | - | - | - | - |
| | **Whey Curd** | - | - | - | 1 | 1 | - | - | | - | - | - | - |
| | **2003** | - | - | 1 | - | - | - | - | - | - | - | - | - |
| | **2003A** | - | - | - | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | - |
| | **2003B** | - | - | - | 1 | 1 | 0 | - | - | - | - | - | - |
| | **2003C** | - | - | - | 1 | 1 | 0 | 0 | - | - | - | - | - |
| | **2009A** | - | - | - | 1 | 1 | 0 | 0 | 0 | - | - | - | - |
| | **2009D** | - | - | - | 0 | 0 | 1 | 1 | 1 | 1 | - | - | - |
| | **2009J** | - | - | - | 1 | 0 | 1 | 1 | 1 | 1 | 1 | - | - |
| | **<u>2011</u>** | - | - | - | - | - | - | - | - | - | - | - | 1 |

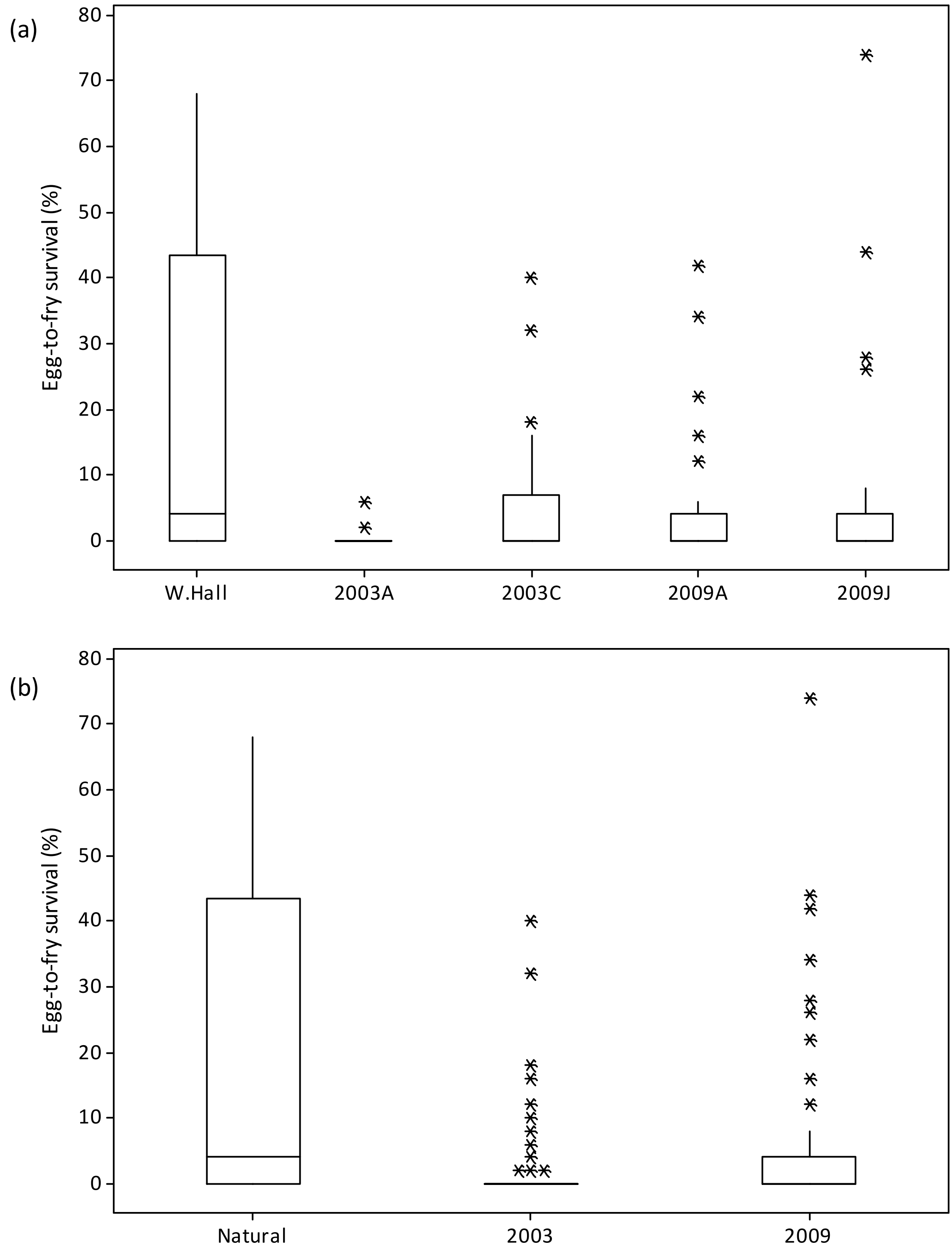



**Figure 5.4** Box plot of egg-to-fry (ETF) survival variability during the 2011 egg-box trial study at site (a), and treatment level (b). Boxes indicate the spread of data between 25-75% of the distribution. The median is marked across the box. Whiskers indicate the full spread of data. * are outliers. Low ETF survival, particularly in 2003 rehabilitation gravel, had many outliers due to the high frequency of egg-boxes with no (zero) ETF survival.

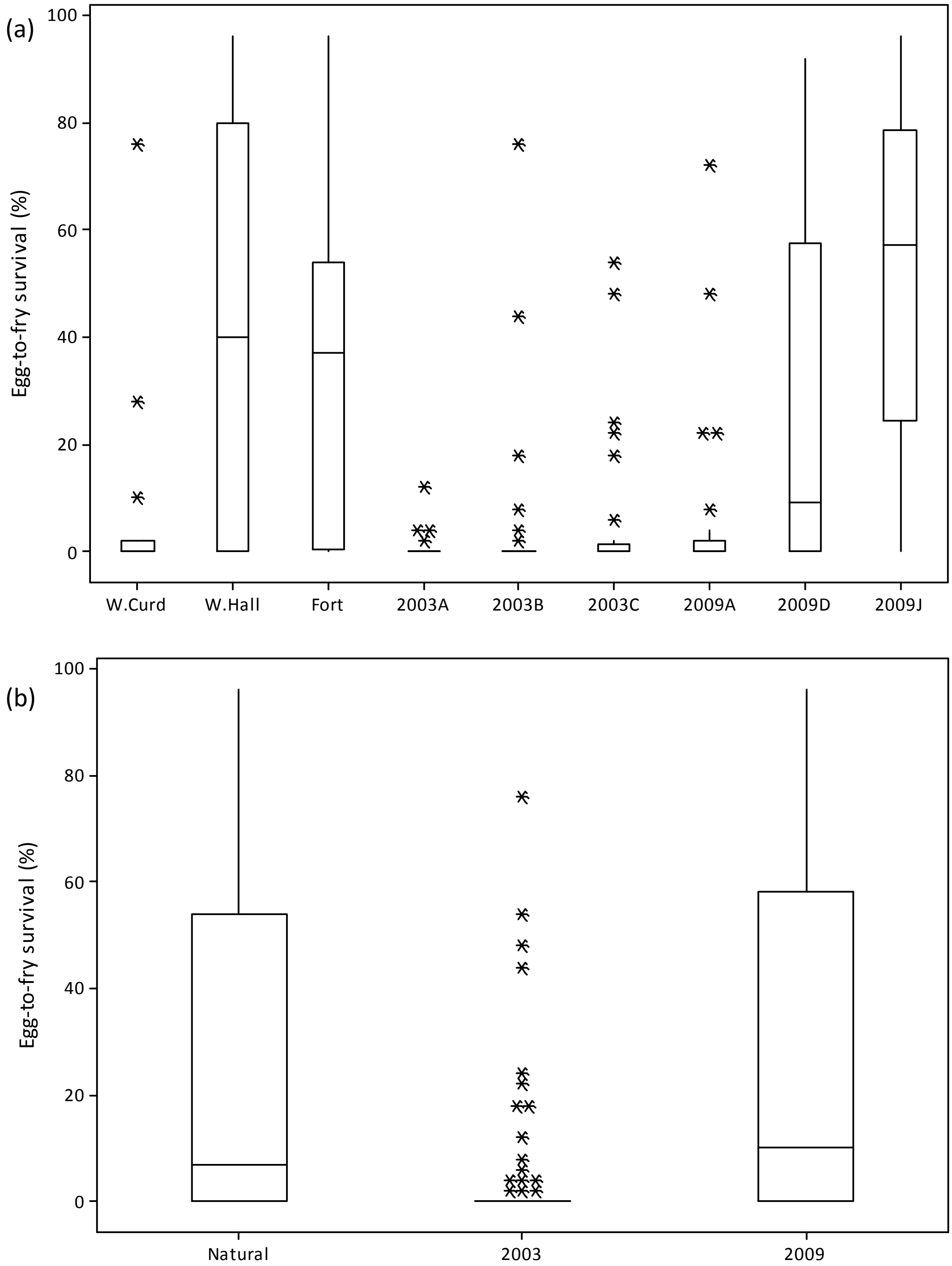
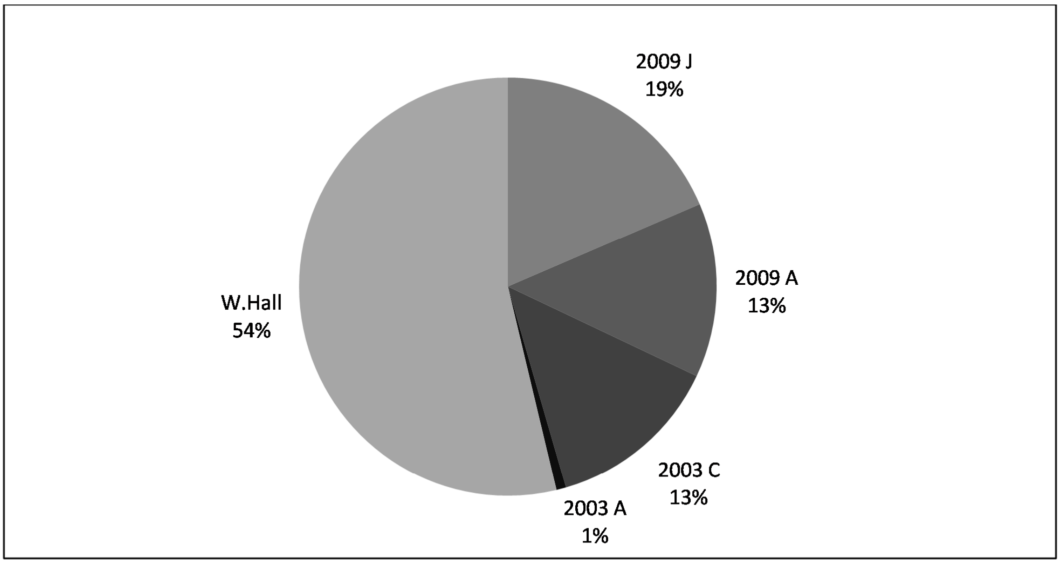



**Figure 5.5** Box plot of egg-to-fry (ETF) survival variability during the 2012 egg-box study at site (a), and treatment level (b). Boxes indicate the spread of data between 25-75% of the distribution. The median is marked across the box. Whiskers indicate the full spread of the data distribution. * are outliers. ETF survival was low, particularly in the 2003 rehabilitation gravel sites, and as such were characterised by many outliers.

(a)




(b)

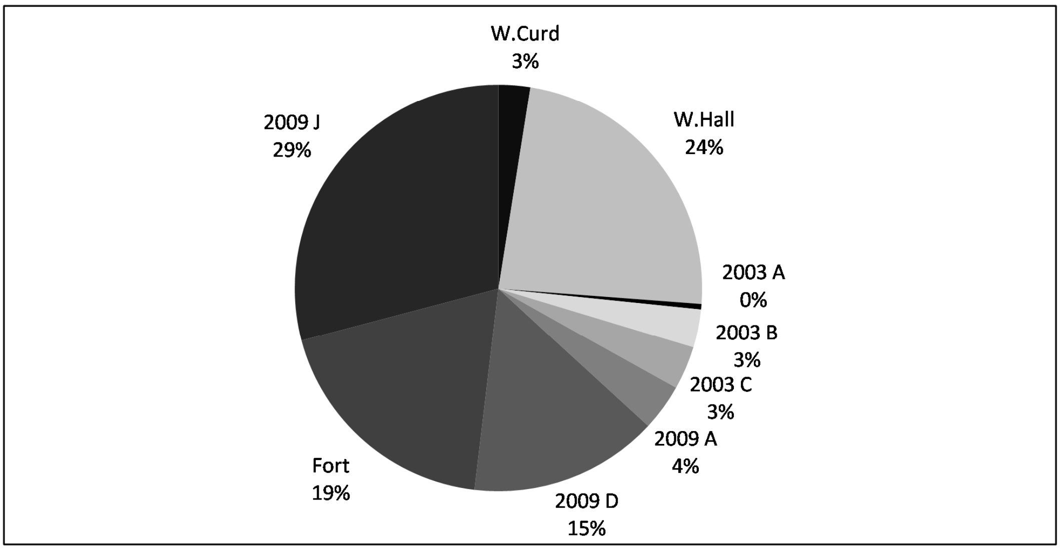



**Figure 5.6** Contributions to cumulative ETF survival (%) per site in 2011 (a) and 2012 (b). The majority of sites contributed little to the cumulative ETF survival total, with most ETF survival observed on only a few sites. The natural gravels at Water Hall had high ETF survival in each year.

## 5.4 Association between egg-to-fry survival, sediment composition and redd velocity

Detrended correspondence analysis (DCA) of cumulative percentage (by weight) gravel ($64>D\geq2$ mm), sand ($2>D\geq0.063$ mm) and silt ($D<0.0063$) from 5-20 cm core depth (the embryo incubation zone) and mean redd velocity from the 2012 study gave a gradient length <3 standard deviation (SD) units for gravel sites (0.896) and treatment (0.297) (Table 5.3). Principal components analysis (PCA) was therefore the more appropriate technique for further analysis because there was suitable agreement with the assumed model of linear response. Axes 1 and 2 cumulatively explained 92.3% of site variation and 96.8% of variation within treatments.

The first principal component of the site biplot explained 79.7% of data variation, and component 2 18.6% (Table 5.4, Figure 5.7). Gravel ($64>D\geq2$ mm) was moderately associated with velocity whilst sand ($2>D\geq0.063$ mm) and silt ($D<0.0063$) were strongly positively correlated (Figure 5.7). Component 1 was driven largely by this sediment grain-size gradient, from gravel to silt/sand. Rehabilitation gravel sites had lower variability and distinct sedimentary compositions. The 2009 rehabilitation gravel treatment had little sediment grain-size variance with greater percentages of gravel distributed along a high to low velocity gradient. The 2003 rehabilitation gravels illustrated greater sediment grain-size variance than the 2009 rehabilitation gravel sites and were distributed along a gravel-sand/silt gradient. The upstream-most 2003 rehabilitation site had the greatest percentage of finer grained sediment, whilst the downstream most site the greatest percentage of gravel. In this manner a spatial pattern of sediment grain-size distribution was evident in the 2003 rehabilitation gravel sites not observed in other gravel treatment sites. Natural gravels were characterised by greater variance in sediment grain-size. ETF survival, plotted as a supplementary variable, had greater association with gravel and velocity.

The first principal component of the treatment biplot accounted for 95.7% data variation, whilst component 2 explained 4.3% of data variation (Table 5.4; Figure 5.7). Similar to the site biplot, there was a positive relationship between silt and sand, whilst gravel and stream velocity were correlated (Figure 5.7). A clear distinction was observed between rehabilitation and natural gravels based on the relationship between sediment gain-size and velocity: rehabilitation gravel had a greater abundance of gravel and experienced higher velocities, whilst natural gravels had greater abundances of sand and silt. ETF survival, however, was moderately associated with the higher percentages of finer grained sediments observed in the natural gravel treatment.

**Table 5.3** Summary table of the Detrended Correspondence Analysis (DCA) of the mean ETF survival (%) response to mean redd velocity and cumulative classified sediment distribution within the embryo incubation zone, 5-20 cm; gravel, sand and silt, for both treatments and sites.

| | **Axes** | **1** | **2** | **3** | **4** | **Total inertia** |
|---|---|---|---|---|---|---|
| **Sites** | Eigenvalues | 0.115 | 0.005 | 0.001 | 0 | 0.139 |
| | Lengths of gradient | 0.896 | 0.303 | 0.181 | 0.56 | |
| | Cumulative percentage variance | 82.8 | 86.1 | 86.6 | 86.8 | |
| **Treatments** | Eigenvalues | 0.053 | 0.003 | 0 | 0 | 0.056 |
| | Lengths of gradient | 0.297 | 0.138 | 0 | 0 | |
| | Cumulative percentage variance | 94.4 | 100.4 | 0 | 0 | |

**Table 5.4** Summary table of variance described by each axis for the PCA ordination. Axes 1 and 2 account for ≥98.3% variation within the data for each of the analyses. See the associated PCA biplot (Figure 5.7) for the association between ETF survival, redd sediment composition and redd velocity.

| | **Axes** | **1** | **2** | **3** | **4** | **Total variance** |
|---|---|---|---|---|---|---|
| **Sites** | Eigenvalues | 0.797 | 0.186 | 0.016 | 0.000 | 1 |
| | Cumulative percentage variance | 79.7 | 98.3 | 100.0 | 100.0 | |
| **Treatments** | Eigenvalues | 0.957 | 0.043 | 0 | 0 | 1 |
| | Cumulative percentage variance | 95.7 | 100 | 0 | 0 | |

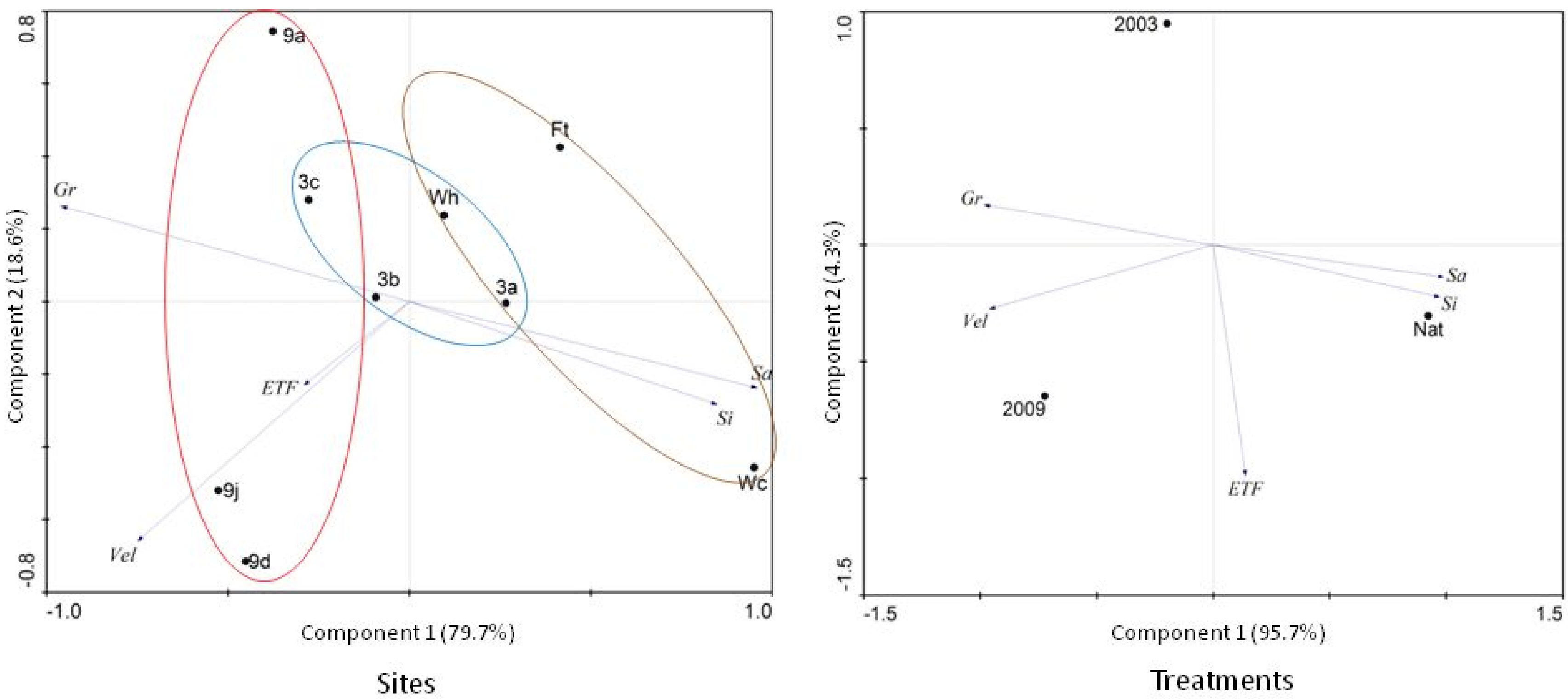



**Figure 5.7** Principal components analysis (PCA) biplot of mean redd velocity and the cumulative classified sediment composition within the embryo incubation zone, 5-20 cm; gravel (Gr), sand (Sa) and silt/clay (Si) for treatments and sites used during the 2012 study only. Rehabilitation site labels indicate the specific treatment and site, for example, '*3a*' refers to site 2003A. Natural sites are coded '*Ft*' for Fort, '*Wc*' for Whey Curd and '*Wh*' for Water Hall. See PCA ordination variance statistics summary Table 5.4.

### 5.5 Sediment composition within redds prior to the introduction of *S. trutta* embryo

Redds cut on the upstream most site within each gravel treatment (2003A, 2009A and Whey Curd) were sampled prior to embryo incubation in order to determine the initial sediment composition of the incubation gravels. Grain-size distributions of incubating substrate within redds differed significantly between treatments prior to embryo installation (Mann-Whitney, $p<0.05$, Table 5.5). However, there was a general tendency across all treatments for the median grain-size diameter ($D_{50}$) to decrease with depth (Table 5.6).

Fine grained sediment (<1 mm) was effectively removed from constituent framework gravels, particularly the upper redd substrate, through the redd cutting process as has been observed naturally (Kondolf et al., 1993; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006; Marchildon et al., 2010) (Figures 5.8 and 5.9). However, loss of sediment <1 mm was not as effective in the natural gravels at Whey Curd >15 cm depth. This was due to the initial high fine sediment (<1 mm) composition at this site. The composition of sediments <1 mm varied significantly between treatments; the redd cut in the natural gravel site, Whey Curd, contained the greatest abundance and the 2009A rehabilitation redd the least (Kruskal-Wallis, $p<0.05$, Table 5.7). This dominance of finer grained sediments was reflected in a low distribution statistic values in subsurface redd substrate (Table 5.6). Moreover, coarse sand ($2>D\geq1$ mm) contributed minimally to the sediment composition of the 2009A rehabilitation gravel red (Figure 5.8).

The sediment composition of the upper substrate within all redds on all gravel treatments consisted mainly of framework gravels ($64>D\geq16$ mm) within a suitable size range for migratory *S. trutta* spawning. The redds cut on rehabilitation gravel contained a large abundance of framework gravels ($64>D\geq16$ mm) throughout the sampled extent. However, gravels within this size range ($64>D\geq16$ mm) rapidly declined below 10 cm within the natural treatment. Furthermore, the redd cut on the natural gravel treatment contained significantly less gravel suitable for non-migratory *S. trutta* spawning ($30>D_{50}\geq16$ mm) than either of the rehabilitation sites (Kruskal-Wallis, $p<0.05$, Table 5.7). Cobbles constituted 26.1% of natural redd substrate weight (Table 5.8). Cobbles used to anchor the 2003 rehabilitation gravels constituted 35.5% of overall weight (Figure. 5.9c and d; Table 5.8). The sorting coefficient increased with depth in the natural gravel treatment, from well sorted surface sediments to very poorly sorted substrate (Figures 5.9 and 5.10). The upper 10 cm of substrate within the redd cut on 2003 rehabilitation gravels were well sorted but became poorly sorted with increased depth, apart from the deepest substrate sampled that was well sorted and consisted of gravels (64>$D$≥16 mm). By contrast, the 2009 rehabilitation gravel sediments were well sorted throughout most of their extent becoming moderately to poorly sorted at 25 cm depth and below only. The larger distribution statistic values of the redd cut on 2009 rehabilitation gravels indicated a high composition of framework gravels and abundant interstitial voids prior to embryo introduction (Table 5.6).

**Table 5.5** Summary results of Kruskal-Wallis and Mann-Whitney U analysis for grain-size distribution difference between pre-incubation redd cores from gravel treatments.

| | Kruskal-Wallis | Mann-Whitney U | |
|---|---|---|---|
| | | **2003** | **2009** |
| **Treatment** | 1 | - | - |
| **Natural** | - | 1 | 1 |
| **2003** | - | - | 1 |

**Table 5.6** Grain-size (mm) dstribution statistics of pre-incubation redd gravel cores based on a single core for each treatment. There was a general tendency for the median grain-size diameter ($D_{50}$) to decrease with depth. Rehabilitation gravel had a greater $D_{10}$ and $D_{50}$ through the mid to lower depth of the core due to a reduced fine grained sediment (<1 mm) composition.

| Treatment | Depth (cm) | $D_{10}$ | $D_{50}$ | $D_{90}$ |
|---|---|---|---|---|
| Natural | 0-5 | 30.37 | 41.10 | 55.63 |
| | 5-10 | 8.94 | 26.94 | 51.02 |
| | 10-15 | 0.46 | 15.69 | 47.74 |
| | 15-20 | 0.03 | 0.55 | 19.06 |
| | 20-25 | 0.01 | 0.27 | 3.61 |
| | 25-30 | 0.00 | 0.24 | 8.96 |
| 2003 | 0-5 | 18.88 | 34.73 | 53.78 |
| | 5-10 | 17.04 | 23.33 | 40.84 |
| | 10-15 | 6.11 | 21.79 | 42.91 |
| | 15-20 | 0.42 | 17.36 | 29.95 |
| | 20-25 | 0.41 | 17.56 | 40.49 |
| | 25-30 | 2.48 | 51.43 | 74.98 |
| 2009 | 0-5 | 17.59 | 33.22 | 53.31 |
| | 5-10 | 16.35 | 21.41 | 28.04 |
| | 10-15 | 16.39 | 22.60 | 37.51 |
| | 15-20 | 9.83 | 19.12 | 27.42 |
| | 20-25 | 7.97 | 13.63 | 25.42 |
| | 25-30 | 0.88 | 14.31 | 37.23 |

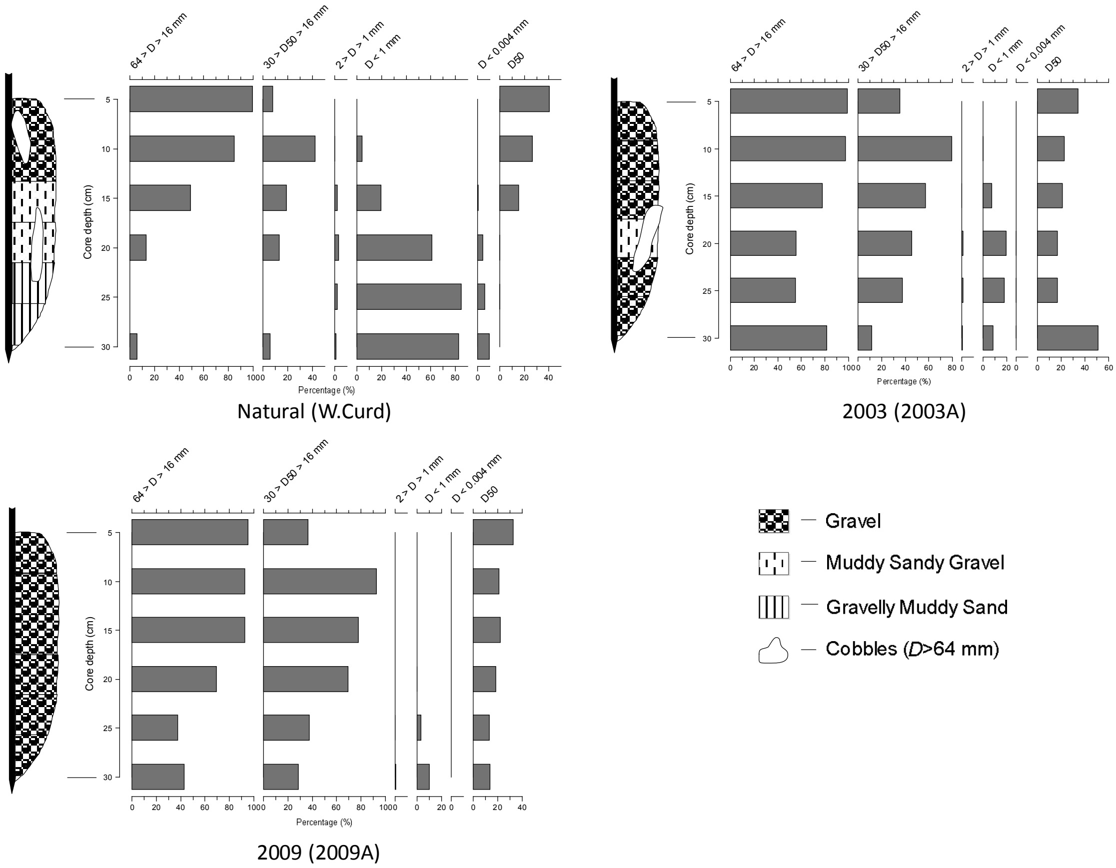



**Figure 5.8** Vertical profile of percentage pre-incubation redd sediment composition and structural characteristics of significance for *S. trutta* spawning from natural gravels at Whey Curd, and rehabilitation gravels at 2003A and 2009A. Redds were cut on the upstream most sites within each gravel treatment during the 2012 study only.

a) b) c) d) e) f)

**Figure 5.9** Examples of pre- and post-incubation redd cores; natural treatment cores extracted from Whey Curd (a) and Fort (b), 2003 rehabilitation gravel cores from 2003A (c) and 2003B (d), and 2009 rehabilitation gravel cores extracted from sites 2009A (e) and 2009J (f). Cores (a), (c) and (e) were pre-incubation, whilst (b), (d) and (f) were post-incubation redd cores. Note the occurrence of cobbles ≥64 mm from the 2003 gravel treatment (c and d), and the abundance of gravel (64>$D$≥16 mm) on the 2009 rehabilitation gravel treatment (e and f).

**Table 5.7** Kruskal-Wallis test results summary for difference in *S. trutta* spawning sediment grain-sizes between pre-incubation redd gravel cores extracted from different treatments.

| Sediment | Treatment | 2003 | Natural |
|---|---|---|---|
| 64>$D$≥16 mm | 2009 | 1 | 1 |
| | Natural | 1 | - |
| 30>$D_{50}$≥16 mm | 2009 | 0 | 1 |
| | Natural | 1 | - |
| 2>$D$≥1 mm | 2009 | 1 | 1 |
| | Natural | 1 | - |
| $D$<1 mm | 2009 | 1 | 1 |
| | Natural | 0 | - |
| $D$<0.004 mm | 2009 | 1 | 1 |
| | Natural | 0 | - |

**Table 5.8** Summary of the cobbles $D$≥64 mm extracted from incubation substrate within the natural and 2003 rehabilitation gravels. No cobbles $D$≥64 mm were extracted from rehabilitation gravel in the 2009A site.

| Site | Depth | Axis (mm) | | | Roundness | g | Cobble% | Core% |
|---|---|---|---|---|---|---|---|---|
| | | a | b | c | | | | |
| W.Curd | 0-10 | 10.9 | 8 | 7.5 | angular | 686.52 | 16.2 | |
| | 10-25 | 10 | 7.5 | 5 | angular | 419.75 | 9.9 | 26.1 |
| 2003A | 10-25 | 14.5 | 11.3 | 9.3 | sub-rounded | 2000 | 35.5 | 35.5 |

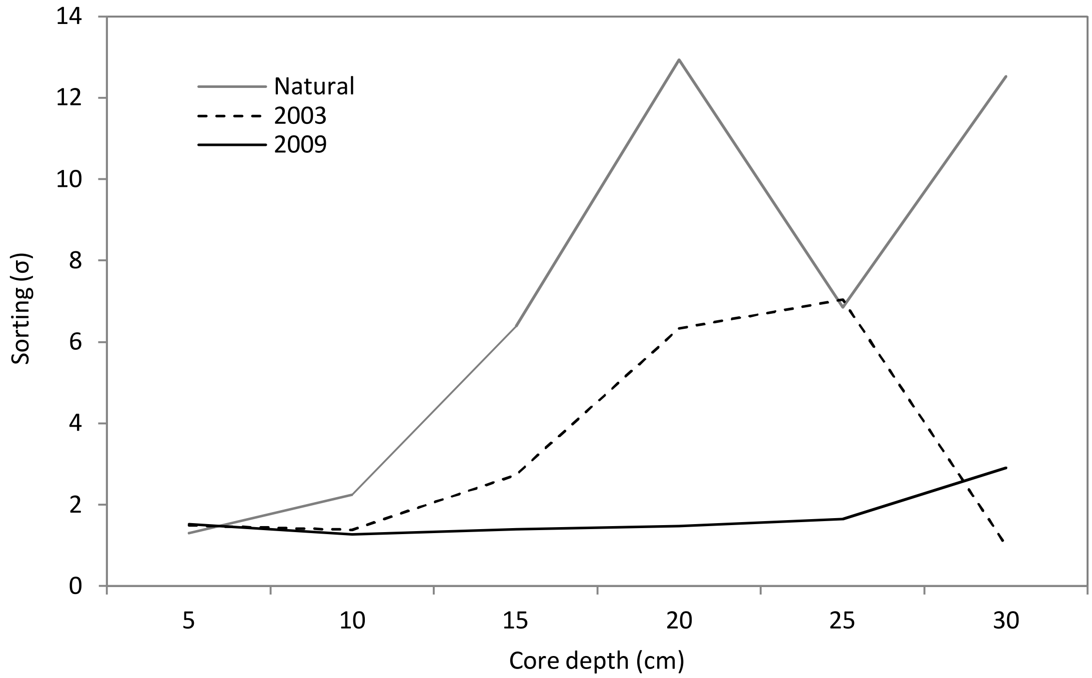



**Figure 5.10** Sorting coefficient (σ) of pre-incubation sediment composition for Whey Curd, 2003A and 2009A gravel sites. Note the high sorting coefficient for the 2009A rehabilitation gravels. The sorting coefficient (σ) was calculated using the Standard Deviation based on the Folk and Ward (1957) method of grain-size distribution sample statistics: $\sigma=\exp(\frac{\ln P16-\ln P84}{4}+\frac{\ln P5-\ln P95}{6.6})$

## 5.6 Redd sediment composition post *S. trutta* embryo incubation

Freeze core samples of redd substrate before and after embryo installation during the 2012 study revealed considerable accrual of sediment ($D$<1 mm and 2>$D$≥1 mm) during the *S. trutta* embryo incubation period. Sediment accumulation changed the composition of the incubating sedimentary environment. Accrual of finer grained sediments ($D$<1 mm and 2>$D$≥1 mm) within spawning substrate during embryo incubation were determined by analysis of the redd sediment composition cut prior to embryo installation and sampled after the incubation period. Post-incubation redds were cut on all 9 sites used during the 2012 study. However, the post-incubation redd on site 2009J could not be identified from the surrounding gravel substrate and as such was not sampled.

Overall, sediment ($D$<1 mm and 2>$D$≥1 mm) ingressed during the embryo incubation period (9 weeks) and settled in deeper interstitial voids, accumulating upwards, in both rehabilitation gravel treatments and the natural gravels on the Whey Curd site (Figures 5.11-5.13). Redds cut on the natural gravels at Whey Curd and at rehabilitation site 2003A had the greatest increase in sediment $D<1$ mm. An upward accumulation of fine sediment ($D<1$ mm) was not observed in natural gravels on either the Water Hall or Fort sites (Figures 5.11). The sediment composition of incubation gravels observed within the redd cut on the Whey Curd site was not characteristic of the natural gravel treatment compared to those from the Fort and Water Hall site (Figure 5.11). Redd substrate recorded at the Whey Curd site were dominated by sediments <1 mm and few spawning gravels ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) below 10 cm depth. Greater abundances of spawning gravel ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) were observed at the Fort and Water Hall site, as well as a relatively less smaller accumulation of sediment (<1 mm). The percentage contribution of gravels ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) in both rehabilitation gravel treatments were, however, greater than in natural gravels.

Cobbles ≥64 mm were observed in the 2003A and 2003B incubation substrata (Figure 5.12) and constituted 45.6% and 51.1% of overall weight respectively. These cobbles were observed in the 2003 rehabilitation gravel treatment only. This was likely due to the erosion of the more mobile grain-sizes from surface sediments and gravel subsidence. However, the sediment composition of redds cut into the 2009 rehabilitation gravel treatment contained the greatest abundance of suitably sized spawning gravels ($64>D\geq16$ mm and $30>D_{50}\geq16$ mm) throughout the vertical extent of the redd and the least accumulated fine sediment <1 mm and $2>D\geq1$ mm (Figure 5.13). Site 2009A had high distribution statistic values throughout the depth of the redd indicating minimal fine grained sediment ($D<1$ mm and $2>D\geq1$ mm) had ingressed during the embryo incubation period (Table 5.9). Sediment within the sand size range ($2>D\geq1$ mm) changed little over the incubation period. The redd cut on the 2009D site experienced greater ingress of sediment $D<1$ mm, however this remained comparatively low across gravel treatments.

Sediment accumulation affected the degree of sediment sorting, mostly within natural gravels and the lower substrata of the 2003 rehabilitation gravel treatment (Figure 5.14). Sediment sorting below the surface substrate was poor for the natural and 2003 rehabilitation gravels. Site 2009A, however, remained in a similar sediment condition to the pre-incubation state, whilst site 2009D became less well sorted at 15-20 cm core depth. Accumulation of sediment ($D<1$ mm and $2>D\geq1$ mm) decreased grain-size distribution $D_{10}$ values of the natural and 2003 rehabilitation gravels (Table 5.9). However, the larger grain-size distribution statistic values within surface substrate of redds cut on rehabilitation gravel were characteristic of a greater percentage framework gravels and interstitial voids compared to natural gravel. The $D_{50}$

decreased within the surface 5 cm of redd substrate in redds cut on natural gravel sites. Further, natural gravels had low, but well dispersed, grain-size distribution statistic values indicating a high percentage of fine sediment (*D*<1 mm and 2>*D*≥1 mm) characterised by few interstitial voids and poor sediment sorting throughout the vertical profile (Table 5.9). The 2003 treatment had a typically lower percentage composition of gravel 64>*D*≥16 mm than redds cut into the 2009 treatment gravels due to greater accrual of sediment *D*<1 mm. A spatial decline of sediment *D*<1 mm and 2>*D*≥1 mm accumulation within incubation substrate was observed; a greater abundance of sediment *D*<1 mm and 2>*D*≥1 mm accrued in the upstream most site, 2003A, whilst the 2003C site accumulated relatively less sediment (*D*<1 mm and 2>*D*≥1 mm) over the embryo incubation period.

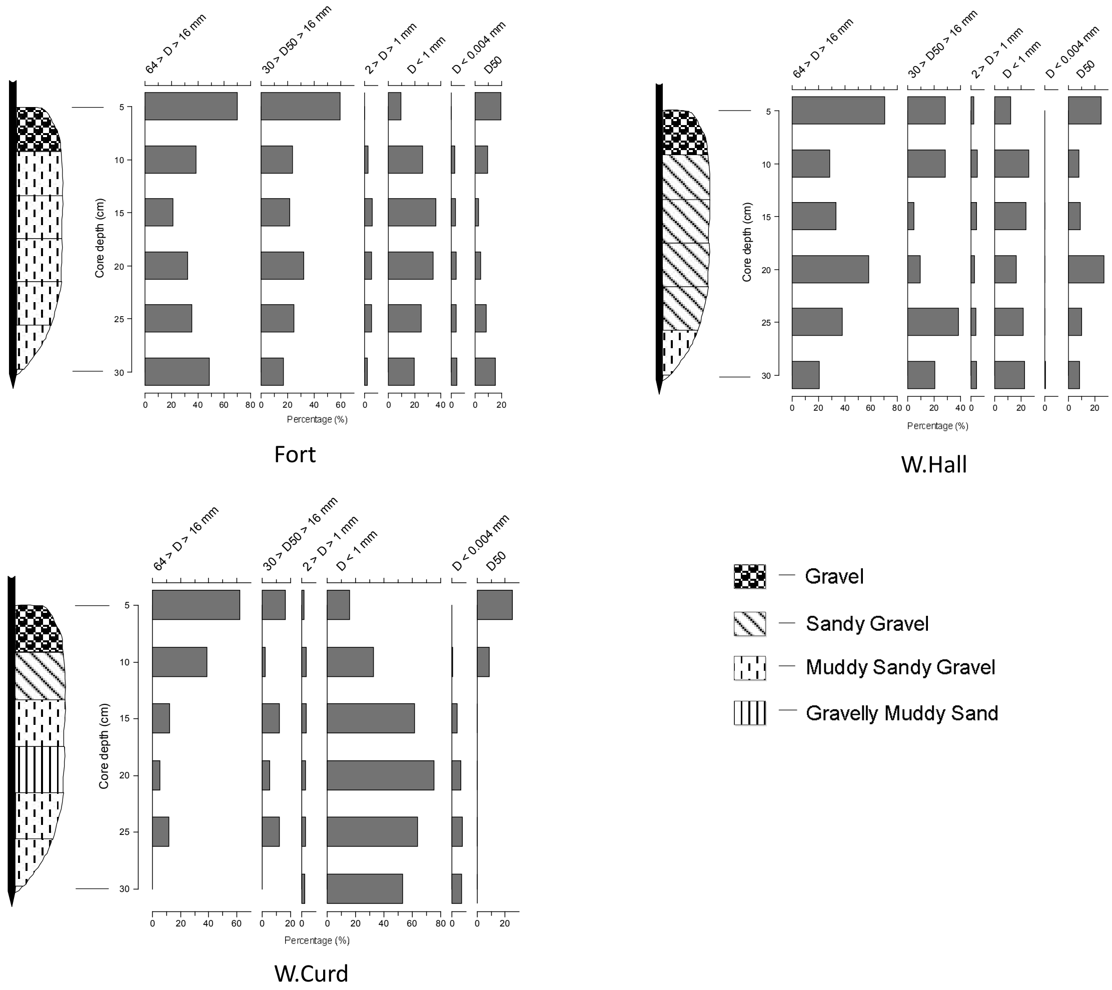



**Figure 5.11** Vertical profile of percentage post-incubation redd sediment composition and structural characteristics of significance for *S. trutta* spawning from all gravel sites within the natural gravel treatment. Note the paucity of gravels 64>*D*≥16 mm and abundance of fine sediment <1 mm in the incubation substrate at the Whey Curd site.

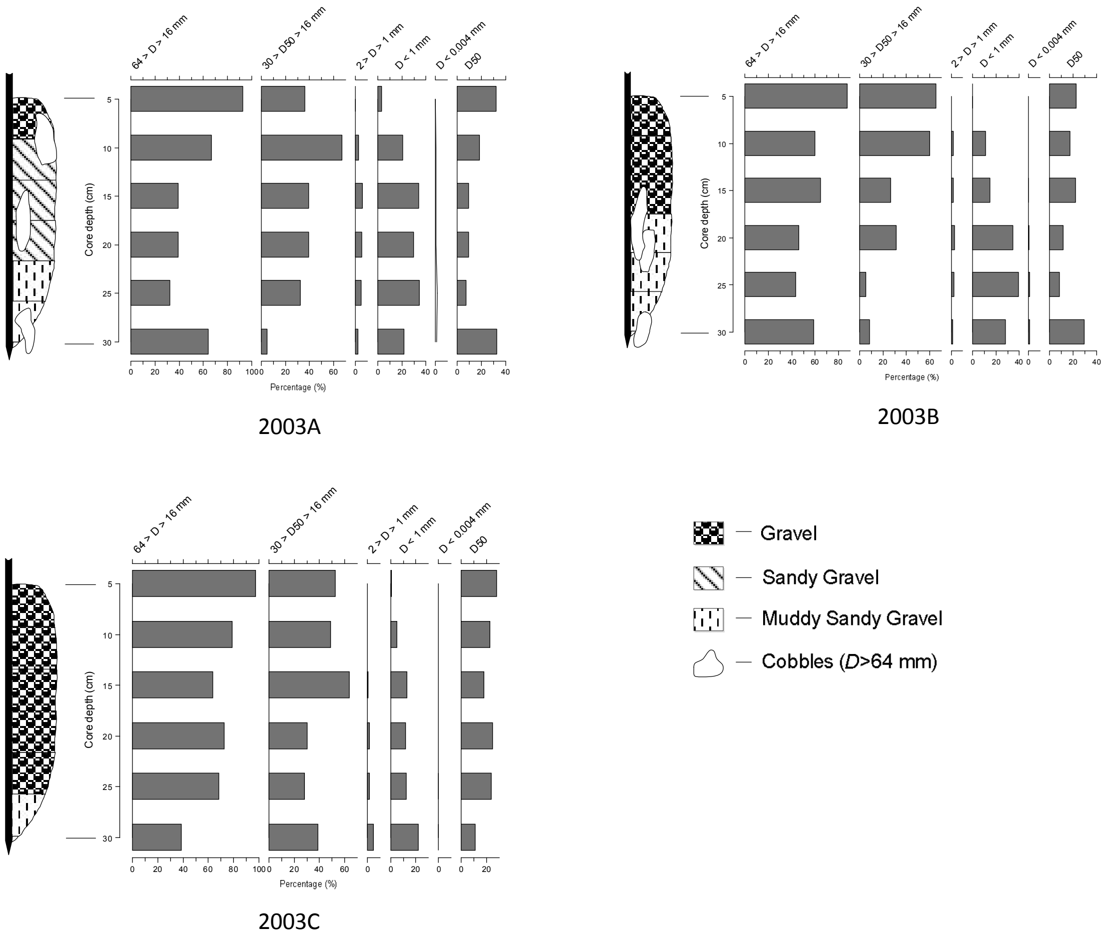
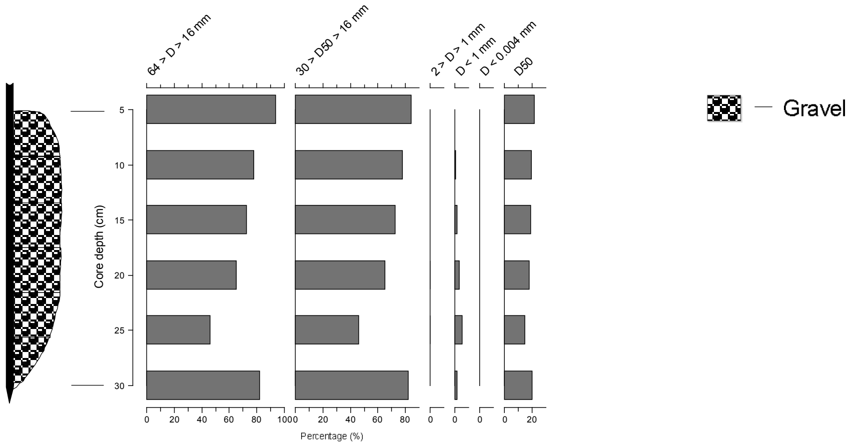
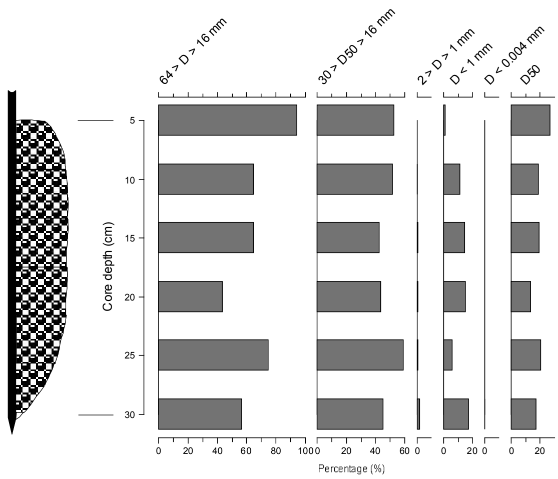



**Figure 5.12** Vertical profile of percentage post-incubation redd sediment composition and structural characteristics of significance for *S. trutta* spawning from all gravel sites within the 2003 rehabilitation gravel treatment. Note the occurrence of cobbles ≥64 mm from sites 2003A and 2003B.




2009A




2009D

**Figure 5.13** Vertical profile of percentage post-incubation redd sediment composition and structural characteristics of significance for *S. trutta* spawning from all gravel sites within the 2009 rehabilitation gravel treatment. Note that the post-incubation redd cut on the 2009J site could not be recovered for sampling as it was not identified from the surrounding gravel substrate.

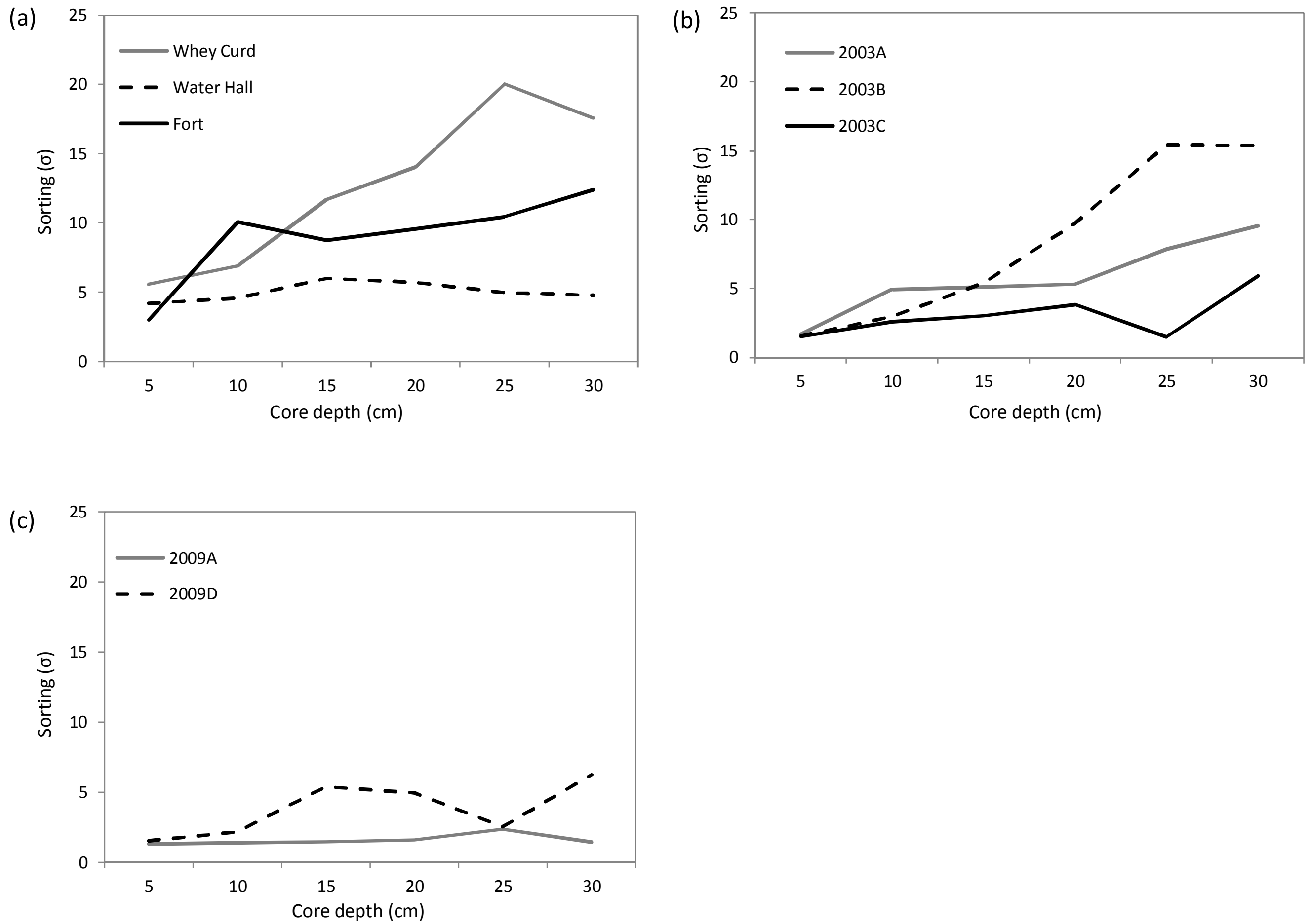



**Figure 5.14** Sorting coefficient (σ) of the post-incubation sediment composition for: natural gravel treatment (a), 2003 rehabilitation gravel treatment (b), and the 2009 rehabilitation gravel treatment (c), based on the Folk and Ward (1957) method of grain-size distribution sample statistics. Note how well sorted the 2009 rehabilitation gravel sites remained during embryo incubation. The post-incubation redd cut into 2009 rehabilitation gravel site 2009J could not be identified and as such was not sampled.

**Table 5.9** Grain-size (mm) statistics of redd substrate post-incubation. Accrual of sediment (*D*<1 mm) decreased grain-size distribution values relative to pre-incubation sediment conditions of the natural and 2003 rehabilitation gravel treatment sites.

| Site | Depth (cm) | $D_{10}$ | $D_{50}$ | $D_{90}$ |
|---|---|---|---|---|
| Fort | 0-5 | 0.83 | 19.85 | 30.91 |
| | 5-10 | 0.25 | 10.19 | 37.80 |
| | 10-15 | 0.28 | 3.32 | 22.53 |
| | 15-20 | 0.25 | 4.51 | 24.76 |
| | 20-25 | 0.18 | 8.74 | 32.25 |
| | 25-30 | 0.05 | 15.61 | 48.36 |
| W.Hall | 0-5 | 0.69 | 25.32 | 50.91 |
| | 5-10 | 0.52 | 8.42 | 24.17 |
| | 10-15 | 0.52 | 9.35 | 46.96 |
| | 15-20 | 0.57 | 27.43 | 52.02 |
| | 20-25 | 0.47 | 10.60 | 25.51 |
| | 25-30 | 0.45 | 8.74 | 22.25 |
| W.Curd | 0-5 | 0.58 | 25.46 | 51.54 |
| | 5-10 | 0.37 | 9.29 | 49.60 |
| | 10-15 | 0.03 | 0.56 | 18.17 |
| | 15-20 | 0.01 | 0.31 | 10.58 |
| | 20-25 | 0.01 | 0.47 | 18.11 |
| | 25-30 | 0.01 | 0.65 | 12.42 |
| 2003A | 0-5 | 16.90 | 32.46 | 53.06 |
| | 5-10 | 0.52 | 18.80 | 27.32 |
| | 10-15 | 0.44 | 9.76 | 25.61 |
| | 15-20 | 0.43 | 10.24 | 25.60 |
| | 20-25 | 0.13 | 7.65 | 24.75 |
| | 25-30 | 0.20 | 33.42 | 53.37 |
| 2003B | 0-5 | 13.86 | 22.91 | 43.61 |
| | 5-10 | 0.76 | 17.83 | 27.03 |
| | 10-15 | 0.59 | 22.58 | 49.93 |
| | 15-20 | 0.10 | 12.25 | 37.58 |
| | 20-25 | 0.05 | 9.08 | 49.94 |
| | 25-30 | 0.05 | 30.11 | 52.27 |
| 2003C | 0-5 | 17.58 | 28.18 | 51.38 |
| | 5-10 | 9.80 | 23.23 | 47.58 |
| | 10-15 | 0.55 | 18.38 | 27.20 |
| | 15-20 | 0.69 | 25.50 | 50.87 |
| | 20-25 | 0.67 | 24.19 | 74.47 |
| | 25-30 | 0.44 | 11.73 | 25.55 |
| 2009A | 0-5 | 16.52 | 22.21 | 29.86 |
| | 5-10 | 10.73 | 20.10 | 27.69 |
| | 10-15 | 9.47 | 19.50 | 27.52 |
| | 15-20 | 8.42 | 18.61 | 27.27 |
| | 20-25 | 6.38 | 15.02 | 26.23 |
| | 25-30 | 8.44 | 20.52 | 27.80 |
| 2009D | 0-5 | 16.91 | 27.20 | 50.82 |
| | 5-10 | 0.57 | 19.23 | 74.92 |
| | 10-15 | 0.46 | 19.99 | 44.02 |
| | 15-20 | 0.40 | 14.07 | 26.00 |
| | 20-25 | 6.75 | 20.84 | 38.28 |
| | 25-30 | 0.50 | 17.59 | 31.98 |

### 5.7 Redd velocity, fine sediment (*D*<1 mm, 2>*D*≥1 mm) and ETF survival

Although redd velocity was low for all sites, Froude numbers indicated subcritical or tranquil stream flow in both 2011 and 2012 (Table 5.10). During the 2011 trial study, mean redd velocities of the natural gravel treatment (Water Hall) were significantly lower (0.44 m $s^{-1}$) than both the 2003 (0.63 m $s^{-1}$) and 2009 (0.66 m $s^{-1}$) rehabilitation gravel treatments (Figure 5.15a and b; $Chi^2$, $p<0.05$, Table 5.11). However, the natural gravel treatment had the greatest ETF survival with >50% cumulative ETF survival (Figure 5.6a). Furthermore, the Water Hall and Fort sites had high ETF survival in the 2012 study, 42.4%, 34.1% respectively (Table 5.1). ETF survival at the Water Hall site was significantly greater than the 2003 rehabilitation gravel sites as well as the 2009A site during both study years (Mann-Whitney, $p<0.05$, Table 5.2).

The Water Hall and Fort sites had significantly greater ETF survival than those cut on the Whey Curd gravels during the 2012 study (Mann-Whitney, $p<0.05$, Table 5.2). The natural treatment also had the greatest difference in sediment (<1 mm) between the pre- and post embryo incubation period (Figure 5.16a), although high sediment <1 mm deposition at the Whey Curd site biased this result somewhat. Moreover, sites within the natural gravel treatment during the 2012 study had significantly lower mean redd velocities (Figure 5.17a and b; $Chi^2$, $p<0.05$, Table 5.11). An increase of coarse sand (2>*D*≥1 mm) within surface sediment (0-10 cm) over the incubation period was also greater in the natural gravel treatment (Figure 5.16b). Consequently, low redd velocities, poor sediment composition of pre-incubation substrate and the high accrual of sediment (*D*<1 mm) likely had a detrimental impact on the incubating embryos at Whey Curd and as such may explain the very low mean ETF survival (4.5%). Redd velocities for the natural gravel treatment did not correlate with ETF survival (Figure 5.18a). However, a repeat of the same test that excluded the natural gravels at Whey Curd, a site of excessive catchment-derived sediment input, yielded a significant but modest correlation (Figure 5.18b; Spearman's Rank-Order, $p<0.05$). This result suggests that excessive sediment input can override the association between redd velocity and ETF survival making it a potential key factor for determining the spawning quality of natural gravels in the River Stiffkey. Greater ETF survival observed at the Water Hall and Fort sites was likely due to the spatially variable nature of spawning sediment composition and structure within individual redds.

Natural treatment spawning gravels, at Whey Curd, during the 2012 study had very high SI scores (a quantitative spawning gravel quality index based on a measure of the composition of sand in spawning substrate) prior to embryo introduction; an indication of a poor embryo incubation environment (Table 5.12).

**Table 5.10** Total mean redd velocity (m $s^{-1}$) for each site and treatment with ± SD and Froude number. Mean redd velocities were low and Froude numbers indicated subcritical or tranquil redd velocity over both years, although total mean redd velocities were greater in 2011. Discharge during egg-box emplacement,3 February 2011, was 0.97 $m^3 s^{-1}$ and embryo recovery, 25 March 2011, 0.63 $m^3 s^{-1}$. Discharge was relatively lower for the 2012 study, 0.30 $m^3 s^{-1}$ and 0.36 $m^3 s^{-1}$ for egg-box emplacement on 11 January 2012 and recovery 19 March 2012 respectively.

| | | | | **Sites** | | | | **Treatments** | | | | **Totals** | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | **Velocity m $s^{-1}$** | | | | **Velocity m $s^{-1}$** | | | | **Velocity m $s^{-1}$** | | | |
| **Year** | **Treatment** | **Site** | | **Mean** | **±** | **SD** | **Fr** | **Mean** | **±** | **SD** | **Fr** | **Mean** | **±** | **SD** | **Fr** |
| **2011** | Natural | W. Hall | | 0.44 | ± | 0.10 | 0.20 | 0.44 | ± | 0.10 | 0.20 | | | | |
| | Old | 2003 A | | 0.60 | ± | 0.16 | 0.28 | | | | | | | | |
| | | 2003 C | | 0.66 | ± | 0.19 | 0.32 | 0.63 | ± | 0.18 | 0.30 | | | | |
| | New | 2009 A | | 0.66 | ± | 0.19 | 0.35 | | | | | | | | |
| | | 2009 J | | 0.66 | ± | 0.24 | 0.34 | 0.66 | ± | 0.22 | 0.35 | 0.60 | ± | 0.20 | 0.30 |
| **2012** | Natural | W. Curd | Jan | 0.28 | ± | 0.13 | 0.16 | | | | | | | | |
| | | | Mar | 0.38 | ± | 0.11 | 0.21 | | | | | | | | |
| | | W. Hall | Jan | 0.33 | ± | 0.14 | 0.19 | | | | | | | | |
| | | | Mar | 0.45 | ± | 0.16 | 0.27 | | | | | | | | |
| | | Fort | Jan | 0.25 | ± | 0.07 | 0.12 | 0.29 | ± | 0.12 | 0.15 | | | | |
| | | | Mar | 0.31 | ± | 0.08 | 0.15 | 0.38 | ± | 0.13 | 0.21 | | | | |
| | Old | 2003 A | Jan | 0.34 | ± | 0.18 | 0.20 | | | | | | | | |
| | | | Mar | 0.46 | ± | 0.13 | 0.28 | | | | | | | | |
| | | 2003 B | Jan | 0.44 | ± | 0.20 | 0.28 | | | | | | | | |
| | | | Mar | 0.53 | ± | 0.15 | 0.34 | | | | | | | | |
| | | 2003 C | Jan | 0.44 | ± | 0.17 | 0.24 | 0.41 | ± | 0.19 | 0.24 | | | | |
| | | | Mar | 0.49 | ± | 0.16 | 0.27 | 0.49 | ± | 0.15 | 0.29 | | | | |
| | New | 2009 A | Jan | 0.37 | ± | 0.13 | 0.21 | | | | | | | | |
| | | | Mar | 0.43 | ± | 0.13 | 0.25 | | | | | | | | |
| | | 2009 D | Jan | 0.61 | ± | 0.22 | 0.42 | | | | | | | | |
| | | | Mar | 0.81 | ± | 0.28 | 0.58 | | | | | | | | |
| | | 2009 J | Jan | 0.56 | ± | 0.26 | 0.38 | 0.51 | ± | 0.23 | 0.33 | 0.40 | ± | 0.21 | 0.23 |
| | | | Mar | 0.82 | ± | 0.28 | 0.56 | 0.69 | ± | 0.30 | 0.45 | 0.52 | ± | 0.24 | 0.31 |

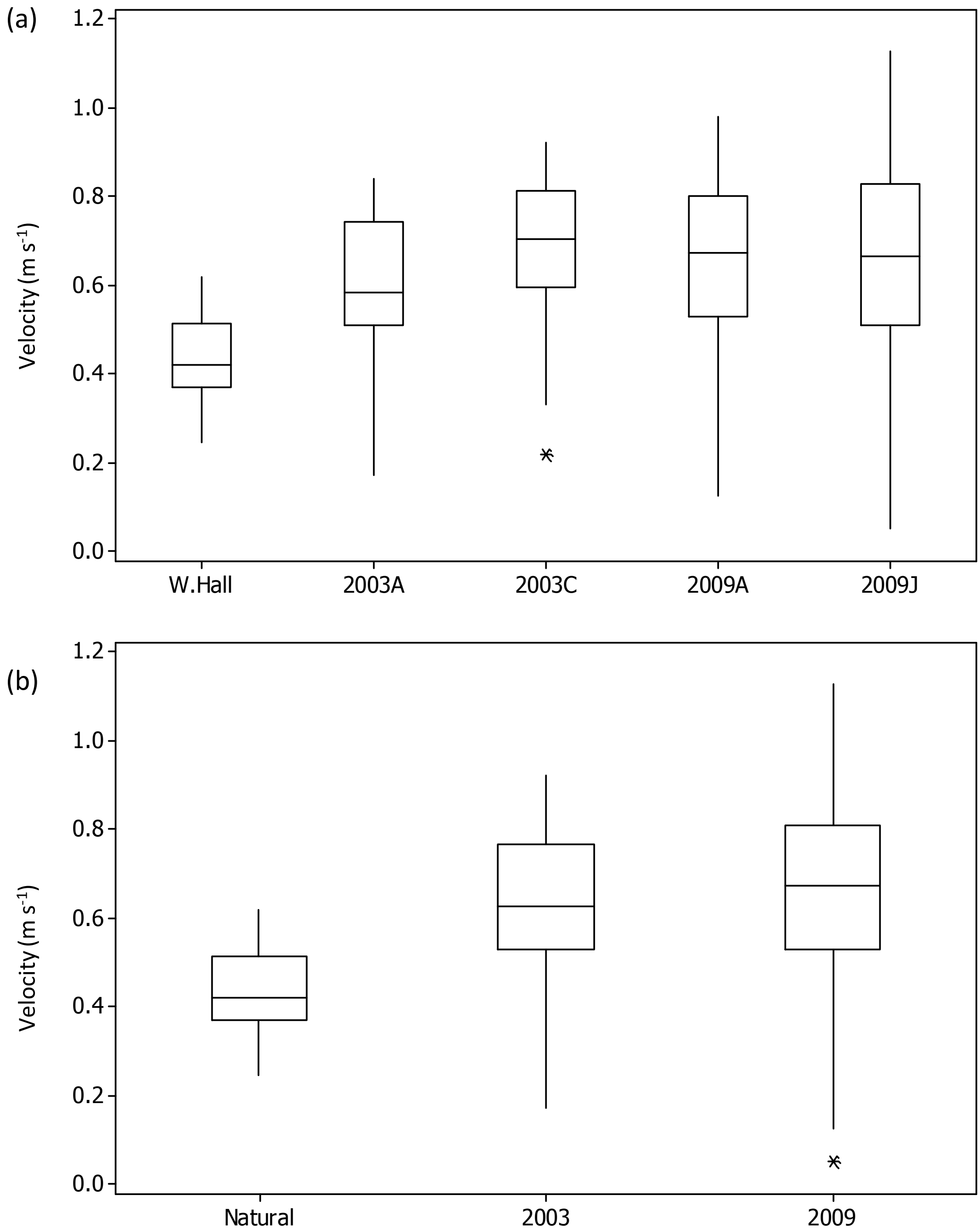



**Figure 5.15** Box plot of mean redd velocity (m $s^{-1}$) measured during March 2011 for gravel sites (a), and treatments (b). Velocity was measured in front of and on both sides of the pit, as well as on the highest point on the tailspill. Boxes indicate the spread of data between 25-75% of the distribution. The median is marked across the box. Whiskers indicate the full spread of the data distribution. * are outliers. Note that redd velocity was only measured post-embryo incubation.

Rehabilitation gravel had a pre-incubation SI score known to be excellent for *S. trutta* emergence (Peterson and Metcalfe, 1981) (Table 5.8). Spawning quality did decrease throughout the incubation period as coarse sand (2>$D$≥1 mm) settled in redd substrate (Figure 5.16b). However, based on its SI index score, site 2009A remained in excellent condition for spawning throughout the incubation period. Furthermore, although sediment <1 mm accumulated within embryo incubation substrate of redds cut on the 2009 rehabilitation gravels, it remained low on site 2009A, increasing to 2.2% of total sediment composition (Figure 5.16a). With a high composition of suitable spawning gravels (64>$D$≥16 mm and 30>$D_{50}$≥16 mm) and low abundance of sediment (<1 mm) (Figure 5.13), the spawning substrate at site 2009A was an ideal habitat for high ETF survival. Although rehabilitation gravel site 2009D had an increase in fine sediment (<1 mm) to 14.4% of the total sediment composition, it had significantly greater ETF survival than 2009A (Figure 5.16a and 5.5a; Mann-Whitney, $p$<0.05, Table 5.2). Moreover, site 2009J had 52.1% ETF survival (Table 5.1). Mean redd velocity, used to deliver oxygen to the developing embryo and remove toxic metabolic waste products, was significantly greater on the 2009 rehabilitation gravels than the other gravel treatments (Figure 5.17b; Chi$^2$, $p$<0.05, Table 5.11). Sites 2009D and 2009J had greater median redd velocities for both January and March, with greater variability and interquartile ranges, than the 2009A site. Although the low redd velocity at site 2009A did not deposit an abundance of finer grained sediment (<1 mm and 2>$D$≥1 mm) within spawning substrate, it is likely that there was not sufficient interstitial gravel velocity and as such embryo survival was limited. Given that sediment conditions were very suitable for high ETF survival, low redd velocity was likely the controlling factor for poor ETF survival at site 2009A.

Given the morphosedimentary succession observed in rehabilitation gravel in the River Stiffkey (see section 4.4, Chapter 4), the significantly poor ETF survival of the older 2003 rehabilitation gravels compared to the 2009 rehabilitation gravel sites (Mann-Whitney, $p$<0.05, Table 5.2) suggests that ETF survival was associated with sedimentary succession. All of the 2003 rehabilitation gravel sites had significantly lower redd velocities during the 2012 study than the 2009 rehabilitation gravel sites (Figure 5.17a; Table 5.10; Chi$^2$, $p$<0.05, Table 5.11) and a largely unsuitable sedimentary environment for successful *S. trutta* embryo development. An abundance of clasts ≥64 mm, and erosion of the smaller gravels (30>$D_{50}$≥16 mm) suitable for non-migratory *S. trutta* spawning (see section 4.4, Chapter 4) made redd cutting appreciably more difficult than on any other gravel treatments site. It is therefore an unsuitable habitat for successful redd building by *S. trutta*. Moreover, accumulation of fine sediment (<1 mm) within the incubation zone (5-20 cm) from pre-incubation levels was 29.1%, 18.9% and 10.5% of total sediment composition for sites 2003A, 2003B and 2003C respectively (Figure 5.16a). A preferential increase of sediment 2>*D*≥1 mm in deeper substrata (Figure 5.16b) contributed

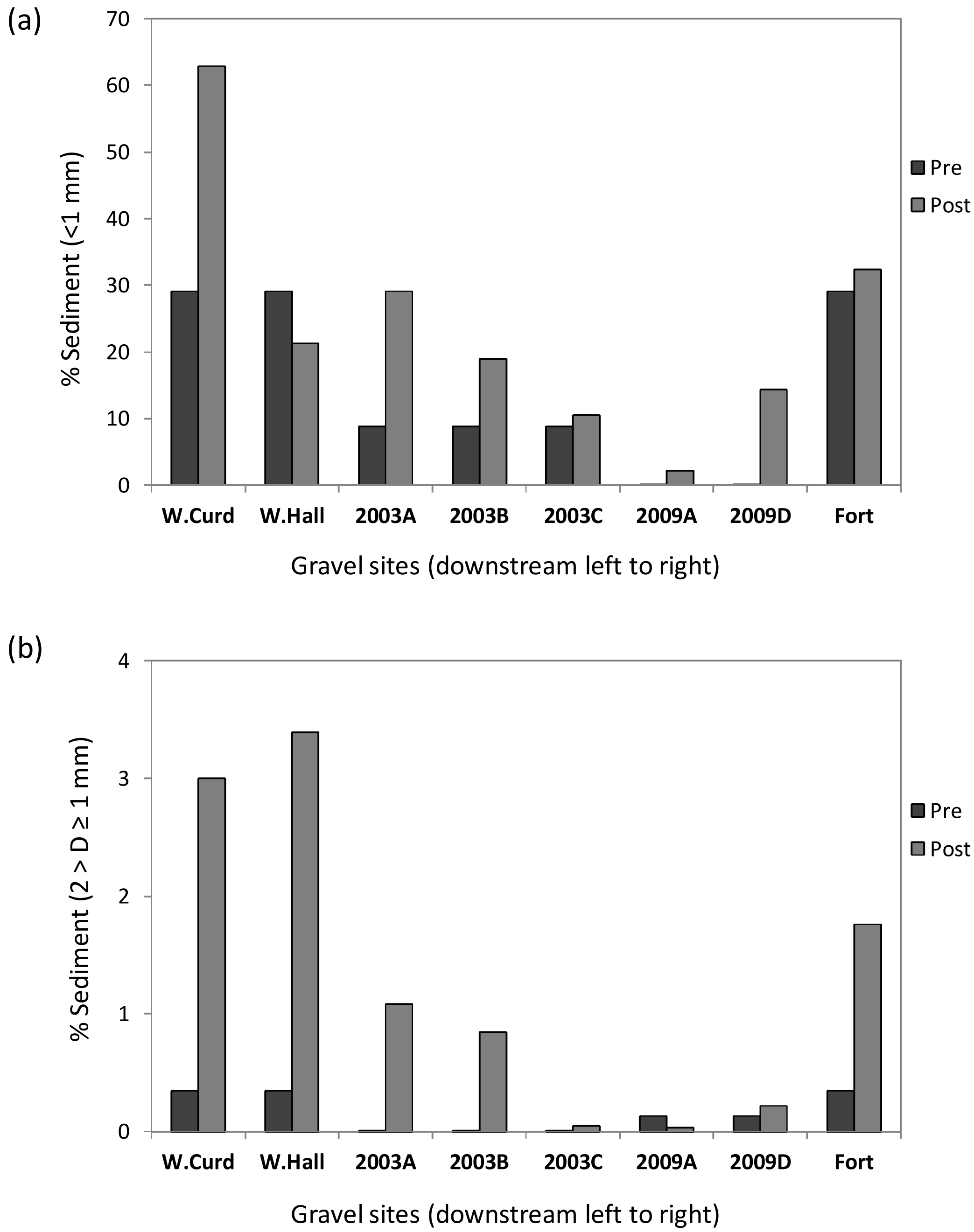



**Figure 5.16** Sediment *D*<1 mm accretion within the incubation zone (5-20 cm) (a), and sediment 2>*D*≥1 mm accretion in surface 10 cm of substrate (b) plotted as cumulative percent sediment accumulation in redd substrate during embryo incubation. Sites are illustrated in a downstream manner left to right.

towards a decline in spawning quality based on SI scores. The more recently installed 2009 rehabilitation gravel treatment sites had significantly greater ETF survival (Mann-Whitney, $p<0.05$, Table 5.2). Moreover, the 2003 rehabilitation treatment had significantly greater median redd velocities than the natural gravel treatment (Figure 5.17b, Table 5.10; $Chi^2$, $p<0.05$, Table 5.11). However, incubation gravels on the natural treatment had significantly greater ETF survival (Mann-Whitney, $p<0.05$, Table 5.2). Given suitable redd velocities for embryo development, the unsuitable sediment composition of pre-incubation substrate on the 2003 rehabilitation gravel treatment limited ETF survival. This reaffirmed the short (<10 years) lifespan of rehabilitation gravel in the River Stiffkey as suitable for *S. trutta* spawning.

**Table 5.11** Chi$^2$ test summary results for difference in redd velocity between treatments within each study year and for both study years combined.

| | | 2003 Treatment | | | Natural Treatment | | |
|---|---|---|---|---|---|---|---|
| **Year** | **Treatment** | **$\chi^2$** | **d.f** | **p-value** | **$\chi^2$** | **d.f** | **p-value** |
| All years | 2009 | 21.88 | 10 | 0.016 | 64.45046 | 10 | <0.001 |
| | Natural | 40.84 | 10 | <0.001 | - | - | - |
| 2011 | 2009 | 9.45 | 10 | 0.490 | 102.80 | 10 | <0.001 |
| | Natural | 83.47 | 10 | <0.001 | - | - | - |
| 2012 | 2009 | 32.01 | 10 | <0.001 | 60.54 | 10 | <0.001 |
| | Natural | 35.04 | 10 | <0.001 | - | - | - |

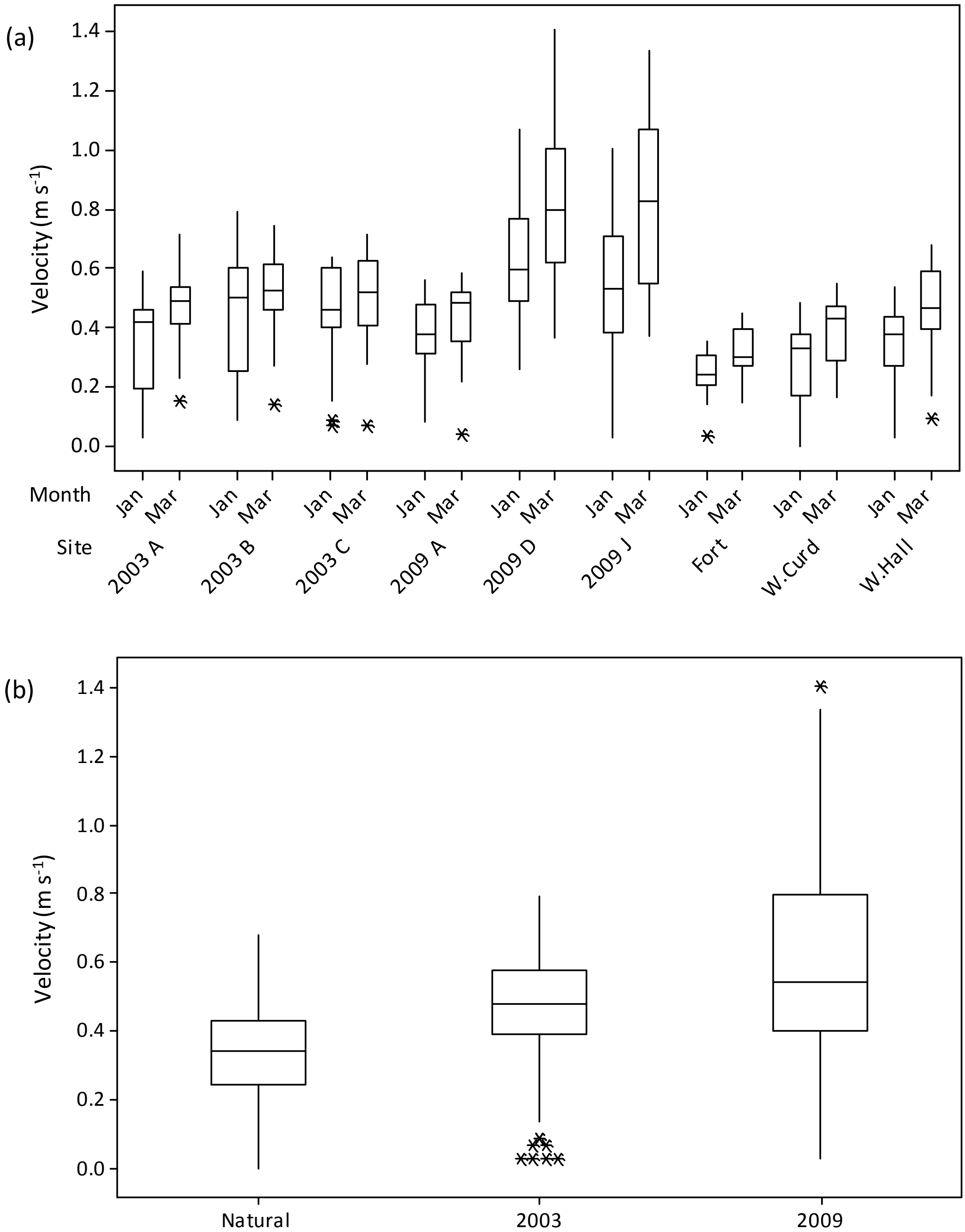



**Figure 5.17** Box plot of mean redd velocity (m $s^{-1}$) measured in January and March 2012 for sites (a), and treatments (b). Boxes indicate the spread of data between 25-75% of the distribution. The median is marked across the box. Whiskers indicate the full spread of the data distribution. * are outliers. Velocity was measured pre- and post-embryo incubation.

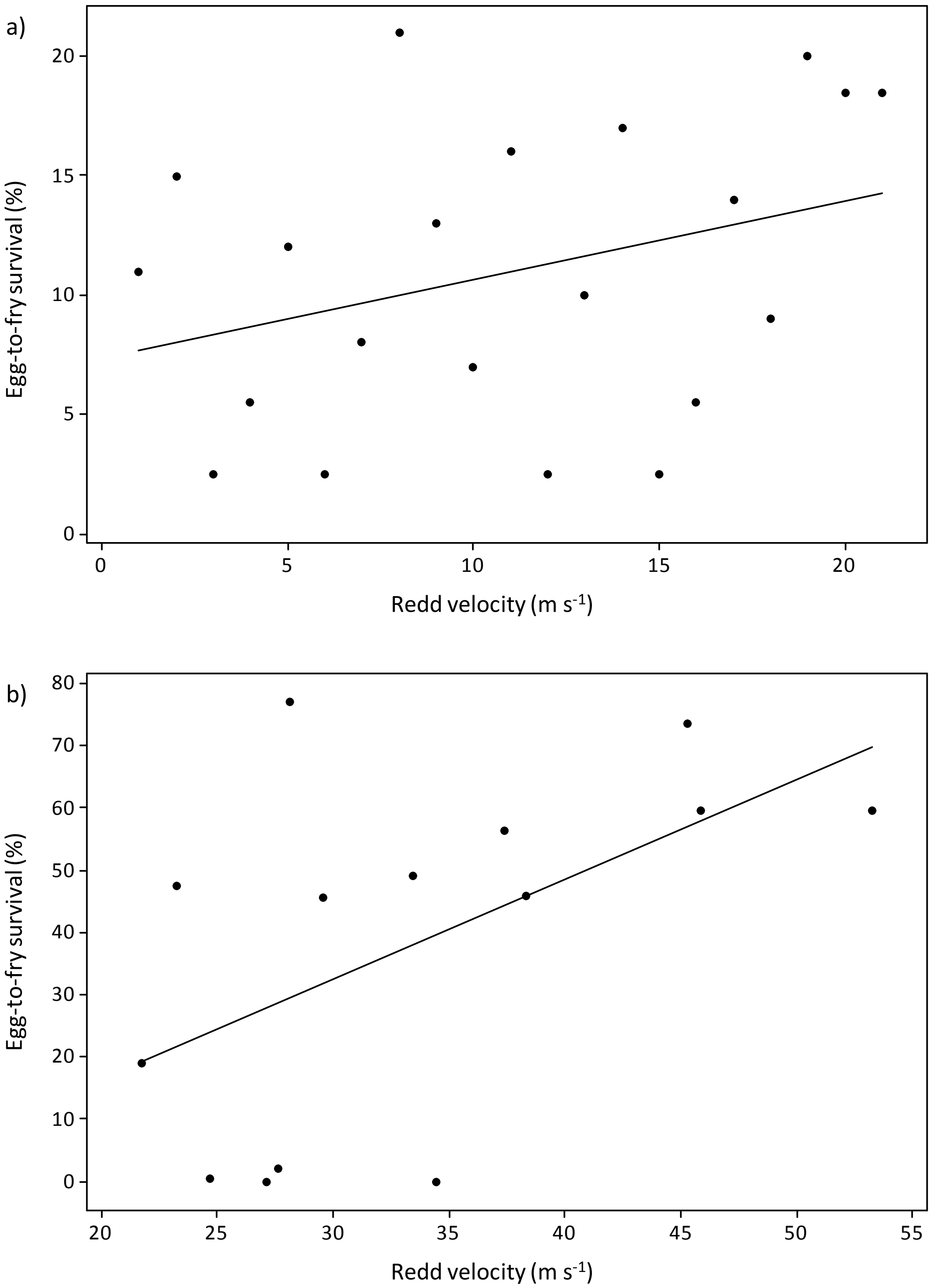



**Figure 5.18** Spearman rank correlation plots of redd velocity and ETF survival for the natural gravel sites, $r_s$=0.331 (a), and redd velocity and ETF survival for Water Hall and Fort sites, $r_s$=0.544 (b) during the 2012 study.

**Table 5.12** Sand Index within the incubating zone (5-20 cm) pre- and post- embryo incubation during 2012. Spawning substrate quality decreased during the embryo incubation period on most sites, except 2009A where it remained in excellent condition for *S. trutta* alevin emergence.

| | **Treatment** | **Site** | **SI** |
|---|---|---|---|
| **Pre** | Natural | W.Curd | 3.21 |
| | 2003 | 2003A | 0.93 |
| | 2009 | 2009A | 0.02 |
| **Post** | Natural | Fort | 3.50 |
| | | W.Hall | 2.09 |
| | | W.Curd | 7.10 |
| | 2003 | 2003A | 2.83 |
| | | 2003B | 1.98 |
| | | 2003C | 1.15 |
| | 2009 | 2009A | 0.27 |
| | | 2009D | 1.58 |

## 5.8 *S. trutta* egg-to-fry survival: effects of catchment-derived sediment on rehabilitation gravels

ETF survival varied significantly between gravel sites and treatments during 2012 (Kruskal Wallis, $p<0.05$, Table 5.2). Variability was associated with ingress of fine sediment (<1 mm) that accumulated throughout the vertical profile of incubation substrata during the embryo incubation period in both rehabilitation and natural gravel treatments (Figure 5.19). The accumulation of sediment (<1 mm) was much greater, and most notable in the upper 0-15 cm of redds cut on natural gravels. Sediment (<1 mm) preferentially settled within deeper substrate in redds cut on rehabilitation gravel. This was likely due to a greater propensity for deeper interstitial voids formed by a high composition of gravels ($64>D\geq16$ mm) and a lower abundance of coarse sand ($2>D\geq1$ mm) within the deeper substrate. Coarse sand ($2>D\geq1$ mm) increased during the embryo incubation period relative to pre-incubation levels at each 5 cm increment of sediment examined for all gravel treatments (Figure 5.20), unlike finer grained sediment (<1 mm). Although coarse sand ($2>D\geq1$ mm) increased in all treatments, the total contribution to overall sediment composition remained below 10%. Both the 2003 and 2009 rehabilitation gravel treatments accumulated far less coarse sand ($2>D\geq1$ mm) in surface sediments than natural gravels, but did however illustrate a preferential settling within deeper substrata in a manner similar to sediment <1 mm. Median percentage increase of coarse sand

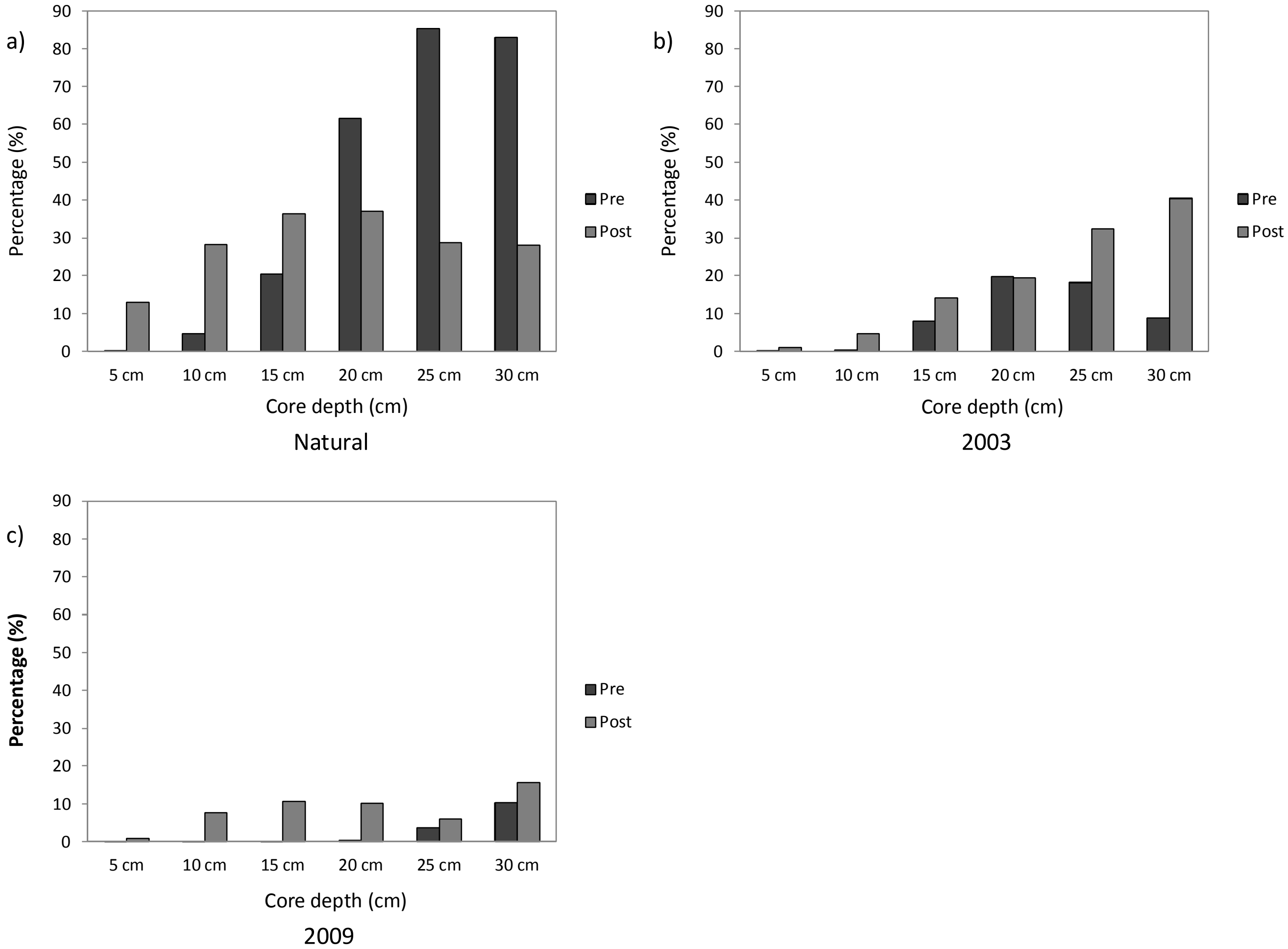



**Figure 5.19** Median percentage fine sediment (*D*<1 mm) to total sediment composition pre- and post-embryo incubation at the natural gravel treatment (a), the 2003 rehabilitation gravel (b) and the 2009 rehabilitation gravel treatment (c). Note how sediment (<1 mm) had a tendency to accumulate in the upper to mid-levels of natural gravels (a), whilst the lower substrate filled preferentially in rehabilitation gravel (b) and (c).

($2>D\geq1$ mm) for all substrate depths at the treatment level however remained below 2.5% of total sediment composition for all gravel treatments. Coarse sand permeated through upper redd substrata and filled deeper interstitial voids preferentially. The SI was high during embryo incubation, and indicated a gradual decline of spawning substrate quality, and a decreased potential for fry to emerge through the upper (0-10 cm) sediment layers (Peterson and Metcalfe, 1981) (Table 5.12).

The fine sediments entering the River Stiffkey were likely eroded and transported from arable fields within the catchment and deposited in the stream channel (see section 3.2.2, Chapter 3). This process was controlled largely by the frequency and magnitude of rainfall. North Norfolk receives most of its annual rainfall during the winter months (Environment Agency, 2005), thus coinciding with the *S. trutta* spawning and embryo incubation period (November to March). Two noteworthy rainfall events, observed in stream discharge (Figure 5.1), occurred during the 2011 embryo incubation period. Two weeks in mid-January received considerable rainfall, with at least a single event of high magnitude ($Q$=1.99 $m^3 s^{-1}$) and short duration forcing the usually moderated chalk stream water levels to rise rapidly. The second event occurred towards the end of February, another high magnitude-short duration rainfall event ($Q$=1.69 $m^3 s^{-1}$). It is likely that suspended sediment loads during these events comprised sediment ($D<1$ mm and $2>D\geq1$ mm) available for deposition within spawning substrate.

A significant reduction in rainfall during the early months of 2012 (Met Office, 2012) reduced river discharge relative to 2011 levels. Redd velocity across all gravel treatments was significantly greater during (March) 2011 than during 2012, averaging 0.6 m $s^{-1}$ (Table 5.10; $Chi^2$, $p<0.05$, Table 5.13). Comparatively, total mean velocity was 0.4 m $s^{-1}$ during embryo installation (January) and 0.52 m $s^{-1}$ during embryo recovery in March 2012. Discharge during 2012 was typically well moderated, reflecting the drier conditions. A single significant rainfall event occurred in early March (Figure 5.1). ETF survival during 2012 was significantly greater than during 2011 (Mann-Whitney, $p<0.05$, Table 5.2), with a total mean ETF survival of 20% in 2012 compared to 7.6% of 2011 (Table 5.1). Both the 2009 rehabilitation gravel and Water Hall site (the only natural gravel site represented in 2011 and 2012) had a significantly higher ETF survival (Mann-Whitney, $p<0.05$, Table 5.2). ETF survival in the 2009 treatment increased from 6.1% in 2011 to 28.8% in 2012, specifically site 2009J, which increased from 7.1% to 52.1% (Table 5.1; Figures 5.4 and 5.5). Good ETF survival was observed on natural gravels at the Water Hall and Fort sites in 2012. The relative increase in ETF survival was associated with a

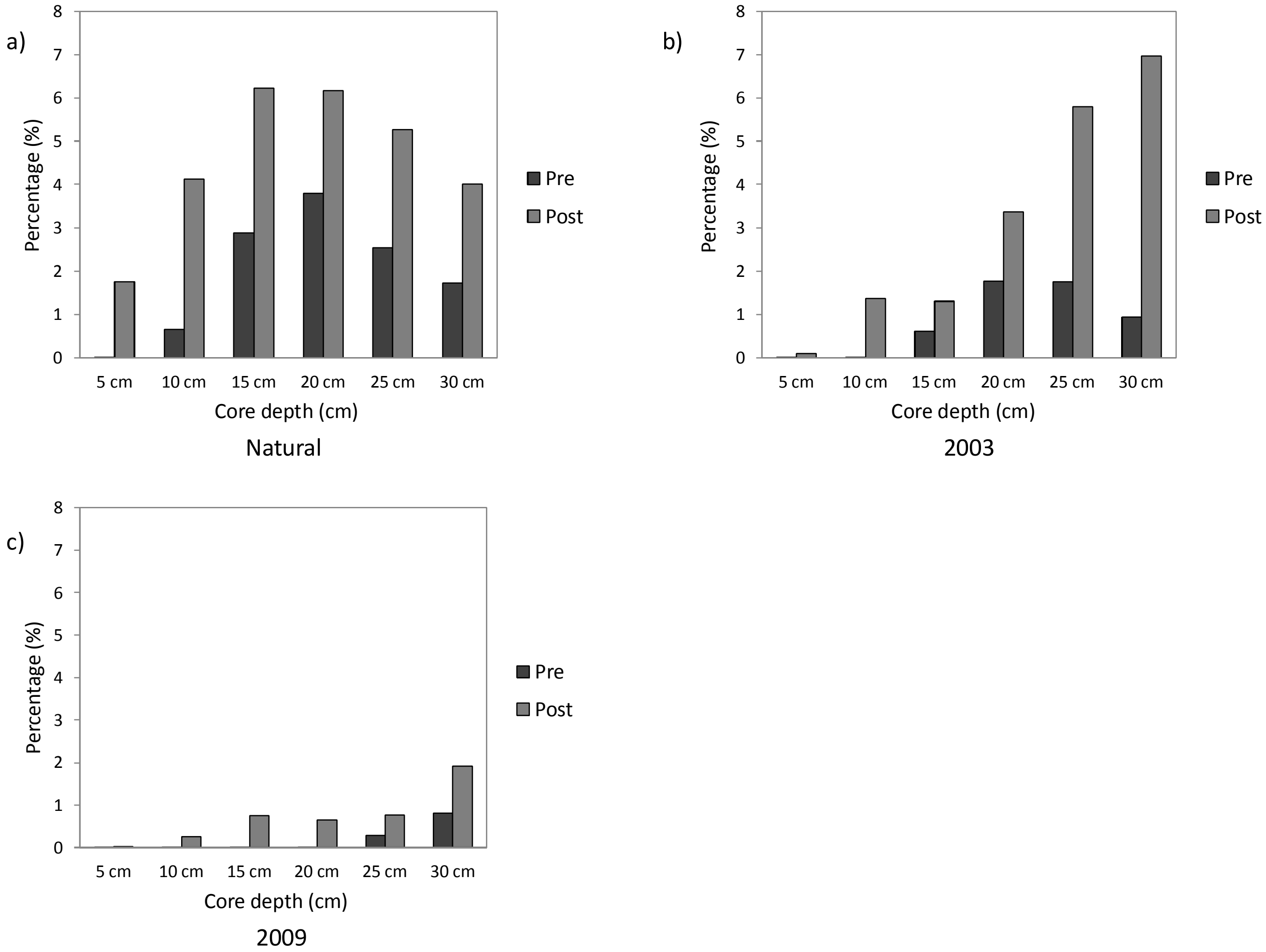



**Figure 5.20** Median percentage sediment 2>*D*≥1 mm of total sediment composition pre- and post-embryo incubation at the natural gravel treatment (a), the 2003 rehabilitation gravel (b) and the 2009 rehabilitation gravel treatment (c). Note how sediment (2>*D*≥1 mm) had a tendency to accumulate in the mid-levels of natural gravels (a), whilst the lower substrate filled preferentially in rehabilitation gravel (b) and (c).

significant decrease in redd velocities between 2011 and 2012 (Table 5.10), which was in turn associated with reduced rainfall (Figure 5.1) and thus catchment-derived sediment pressure. In this way, higher ETF survival in 2012 may be indicative of reduced sedimentation pressures on spawning sediments. However, no significant difference was observed in the 2003 rehabilitation gravels for these periods, suggesting that this treatment did not respond to the potential sediment-load stress reduction in 2012. It is likely that that these treatment sites had only a minor *S. trutta* spawning and embryo development function and contributed little to population recruitment in the River Stiffkey.

Natural gravels at the Water Hall and Fort sites had significantly greater ETF survival compared to the gravels at Whey Curd (Mann-Whitney, $p<0.05$, Table 5.2). Elevated levels of sediment ($D<1$ mm and $2>D\geq1$ mm) were observed at the Whey Curd site (Figure 5.16a and b). The cumulative percentage of fine sediment ($D<1$ mm) within the embryo incubation substrate in the Whey Curd site prior to embryo introduction was prohibitively high for successful development of *S. trutta* embryos (Peterson and Metcalfe, 1981), more than double the 14% threshold limit for 50% emergence (29%). During the embryo incubation period, fine sediment <1 mm more than doubled in this site to 62.9%, whilst accumulations of sediment <1 mm at Water Hall (21.3%) and Fort (32.4%) suggest that fine sediment supply was less of a concern at these sites. It is a likely that furrows cut through the riparian buffer strip of an arable field at the Whey Curd site facilitated sediment laden run-off into the channel where large quantities of fine grained sediment were deposited into spawning substrate. Additionally, the Wighton village road bridge was identified as a point source where large volumes of sediment-laden run-off entered the river channel directly upstream of site 2003A (see section 3.2.2, Chapter 3). The input of sediment ($D<1$ mm and $2>D\geq1$ mm) had an observable spatial distribution within the spawning substrate, decreasing with distance downstream (Figures 5.16). For example, site 2003A accumulated 20.3% sediment ($D<1$ mm), site 2003B 10.1% and site 2003C just 1.7% above pre-incubation sediment levels. A spatial pattern of increasing ETF survival was associated with sequentially lower percentages of sediment ($D<1$ mm) deposition within spawning substrate at each of the successive downstream sites from the Wighton road bridge; 2003A, 2003B, 2003C and 2009A in 2011 and 2012 (Figure 5.21). In a similar manner, the sudden increase of sediment ($D<1$ mm and $2>D\geq1$ mm) in the 2009 rehabilitation gravel site 2009D as well as in the natural gravels at the Fort site are likely further inputs of catchment-derived sediment (Figure 5.16a and b). Catchment-derived sediment influx into the River Stiffkey affected the suitability of rehabilitation gravel by altering sediment composition with consequent impacts on ETF survival.

**Table 5.13** Chi$^2$ test summary results of annual difference in redd velocity between gravel treatments.

| Year | Treatment | 2012 | | |
|---|---|---|---|---|
| | | $\chi^2$ | d.f | p-value |
| **2011** | Natural | 40.03 | 10 | <0.001 |
| | 2003 | 55.47 | 10 | <0.001 |
| | 2009 | 38.680 | 10 | <0.001 |

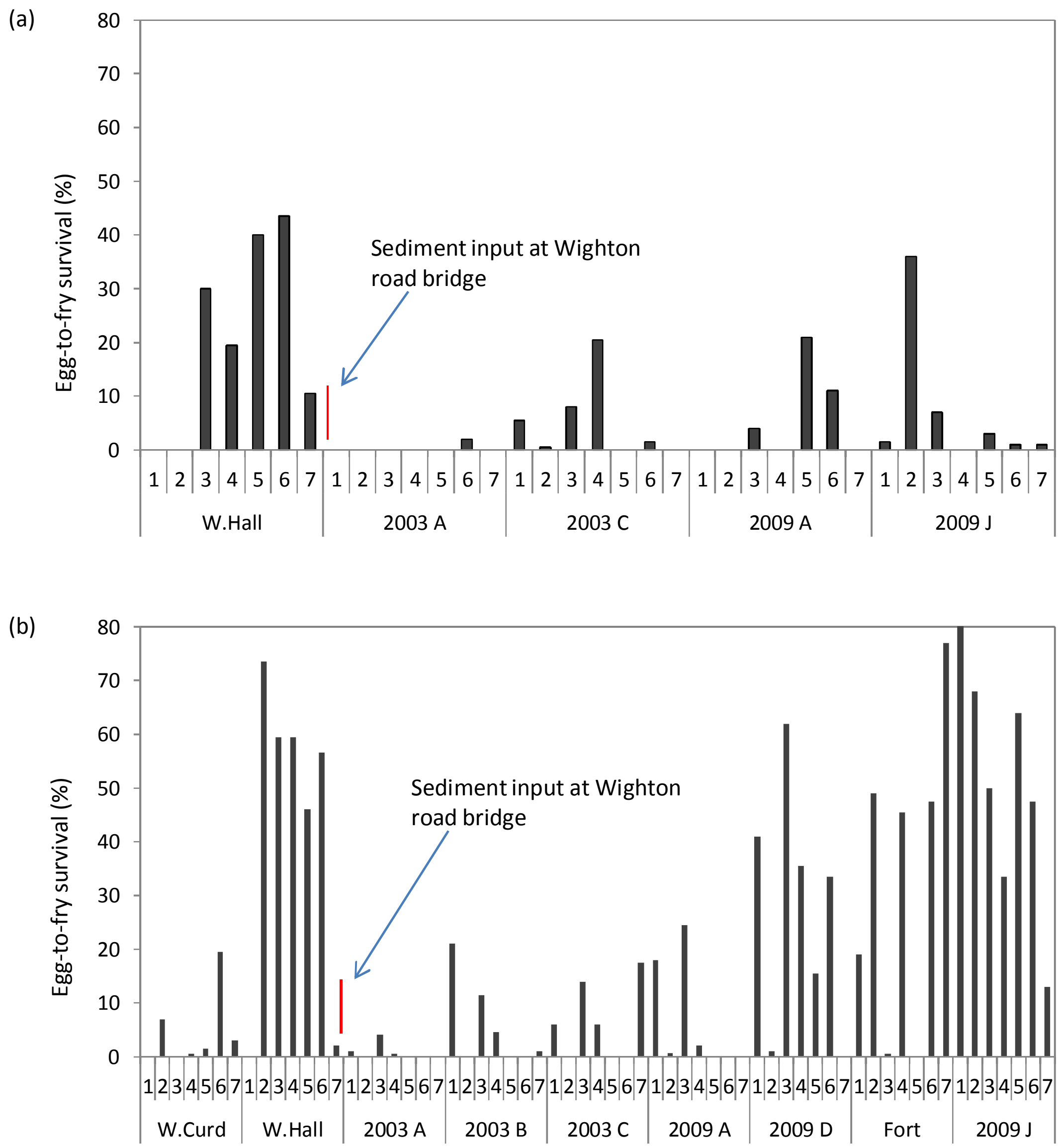



**Figure 5.21** Egg-to-fry (ETF) (%) survival in 2011 (a) and 2012 (b). Redds are numbered 1-7 within each site, which are illustrated in a downstream manner left to right. Note the similar spatial distribution of ETF survival in 2011 and 2012 associated with catchment-derived sediment input.

## 5.9 Discussion

### 5.9.1 Controls on egg-to-fry survival: implications of sediment ingress

Installing gravels as part of rehabilitation schemes acts to supplement naturally occurring habitat and has been shown to have significant impacts on *S. trutta* embryo survival rates (Barlaup et al., 2008; Pulg et al., 2013) and subsequently population recruitment (Pulg et al., 2013). However, constraints to the success of gravel rehabilitation for *S. trutta* spawning vary between river systems and catchment areas, and as such the transient nature of rehabilitation gravel to provide a sustainable habitat for *S. trutta* recruitment is often associated with site specific factors. In rivers not effected by excessive fine sediment input, rehabilitation gravel can lead to permanent ecological improvements (Barlaup et al., 2008) without the need for further management intervention such as gravel jetting. It is often the case, however, that localised habitat improvement works, such as the addition of rehabilitation gravel to streambeds, do not address the cause/s of habitat degradation (Hendry et al., 2003).

Egg-to-fry (ETF) survival, encompassing both embryonic and larval fish life-stage, is a good biological indicator of spawning habitat quality (Dumas and Marty, 2006). Quantification of ETF survival using the egg-box method outlined in Harris (1973) was suitable for use on rehabilitation gravel in the River Stiffkey. As egg-boxes accumulate sediment at the prevailing rate, this method was comparable to installing eggs directly into redd gravel without any form of containment (Harris, 1973), with the added advantage of ETF quantification. Further, there is little disturbance to gravel composition and structure during egg-box installation and as such approximates natural survival well (Dumas and Marty, 2006).

The natural spawning process reduces the fine sediment content of incubation gravels (Kondolf et al., 1993) and fine sediment infiltration during the incubation period has detrimental knock-on implications for embryo development. As the median grain-size ($D_{50}$) of gravel within incubation substrate decreases in response to fine grained sediment, there is typically a decrease in ETF survival (Theurer et al., 1998; Greig et al., 2005a; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006). The mechanism by which this transpires may be direct or indirect, and dependent on sediment composition. For example, clay (0.004 mm) can directly affect ETF survival. Greig et al., (2005b) noted that clay (<0.004 mm) formed a layer around the egg membrane, blocking pores and prevented efficient exchange of DO with ambient water. Due to this, accrual of clay during the incubation period can have disproportionate impacts on ETF survival; 0.3-0.5 g of clay was reported to reduce embryo oxygen consumption by 40-98% (Greig et al., 2005b). Sediment accumulation in incubation gravels is not necessarily the primary controlling mechanism of ETF survival however. Accrual of sediment can block gravel interstices, reducing permeability and consequently the delivery of DO through inhibited interstitial water flow, affecting ETF survival indirectly. In this manner gravel permeability, interstitial velocities, biological oxygen demand (BOD), upwelling of groundwater, and DO concentrations within incubation gravels control the concentration of DO available for incubating embryos (Greig et al., 2007).

### 5.9.2 Sediment intrusion, dissolved oxygen and ETF survival

Although 75% of variation in ETF survival during the 2012 study was associated with the relationship between gravel ($64>D\geq2$ mm) and redd velocity (Figure 5.7), no indirect correlation with accumulated fine grained sediment (<1 mm) was observed. In a similar investigation into ETF survival, Greig et al. (2005a) installed eyed *S. salar* (Atlantic salmon) eggs into redds artificially cut into natural gravels on four salmonid rivers: the Rivers Test (at Horsebridge) and Blackwater in Hampshire, England, and the Rivers Ithon and Aran in Powys, Wales. Although ETF survival was highly variably between sites within each river, the authors found no direct correlation between ETF survival and fine sediment accrual, concluding instead that poor ETF survival was directly correlated with low DO concentrations caused by fine sediment deposition. However, in their review of published literature Jensen et al. (2009) found predictive relationships between percent fine sediment accrual and ETF survival of several salmonid species in North America: Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*) and steelhead (*O. mykiss*). On average, Jensen et al. (2009) found that 1% increase in fine sediment (<0.85 mm) resulted in an approximate 17% ETF survival reduction in all salmonid species examined. Further, ETF survival increased by >20% as the median grain-size ($D_{50}$) increased from 5 to 5.25 mm, and by >10% if $D_{50}$ increased from 10 to 10.25 mm. Using a homogonous gravel mix under laboratory conditions, Olsson and Persson (1986) tested individual sized gravel within the range $1.5\geq D\geq32$ mm and reported a direct correlation between gravel size and *S. trutta* ETF survival, with the greatest ETF survival (95%) resulting from gravel $D$=18 mm. In their field studies, Syrjänen et al. (2008) noted that high ETF survival (83-98%) was not that correlated with the accumulation of fine sediment <1 mm, although this was low and water quality and DO concentrations were high. Comparatively very high ETF survival (93-98%) was reported from incubation substrate within the range $30>D>120$ mm, whilst finer substrate (<4 mm) had poor ETF survival, <40% (Rubin, 1995).

Further evidence of the detrimental effects of fine sediment on developing salmonid embryos was observed in a study conducted by Lapointe et al. (2004). These results indicated that low concentrations (<0.5%) of particles <0.063 significantly impacted ETF survival. Rubin (1998) concluded that ETF survival was associated with levels of interstitial DO that was dependent on substrate permeability, which is in turn determined by embryo incubation grain-size composition.

Interstitial oxygen concentration is typically directly associated with embryo survival, is positively correlated with gravel permeability, and indeed an increase in incubation substrate $D_{50}$ (Maret et al., 1993; Ruben and Glimsäter, 1996; Ingendahl, 2001). Rubin and Glimsäter (1996) concluded from their *S. trutta* egg-box study in a Gotland stream, Denmark, that an ETF survival >50% was correlated with a $D_{50}$ threshold of ≥15 mm through increased permeability and interstitial oxygen concentration. ETF survival of the natural gravels in the River Stiffkey had a similar response to grain-size $D_{50}$; the Whey Curd site had low $D_{50}$ values and very poor ETF survival, whilst the Water Hall and Fort sites had greater grain-size $D_{50}$ values and associated ETF survival. An abundance of sediments with coarser grain-sizes including cobbles ≥64 mm within incubation substrate on the 2003 rehabilitation gravel treatment led to larger $D_{50}$ values than observed in natural gravels as well as the 2009 rehabilitation treatment. However, the sediment structure and composition of these gravel sites were not suitable for *S. trutta* incubation and as such the 2003 rehabilitation gavels were characterised by very poor embryo survival. The Olsson and Persson (1986) tests on individual gravel sizes did not determine the implication for embryo survival of larger gravel sizes, but given the greater interstitial voids inherent in gravel mixes of larger gravel sizes they would likely have had similar results.

An egg-box study, conducted in the River Ebbw Fawr, south Wales, examined the implications on embryo development caused by excessive sediment accrual (Turnpenny and Williams, 1980). 200 eyed rainbow trout (*Oncorhynchus mykiss*) eggs contained in mesh egg-boxes were installed into two natural incubation gravel treatments, one with a high suspended sediment load and another with low suspended sediment. Results were very symptomatic; high suspended sediment load was associated with low ETF survival. The authors linked poor ETF survival to a progressive decline in DO concentration throughout the incubation period and concluded that sediment intrusion decreased gravel permeability lowering interstitial water velocity and subsequently DO concentration. Moreover, agriculturally dominated catchments are frequently associated with a high abundance of sediment-bound nutrients. The interstitial microbial communities that develop in sediment deposits increase biological oxygen demand (BOD) within spawning gravels (Greig et al., 2005a; Greig et al., 2007).

### 5.9.3 The effects of groundwater on ETF survival

Although ETF survival in the River Stiffkey was poor to moderate, ETF survival in natural gravels compared favourably with other studies of chalk streams. Greig et al. (2005a) reported that mean ETF survival on the River Test in Hampshire, England was very low (8.7%). Although there is a direct correlation between DO and ETF survival in groundwater dominated streams (Sowden and Power, 1985; Malcolm et al., 2003), ETF survival may not necessarily be correlated with sediment accrual. Groundwater DO concentration is independent of the incubation sediment composition and it is probable, therefore, that it has a key role in determining embryo survival success in these streams (Sowden and Power, 1985; Greig et al., 2005a). Malcolm et al. (2003) correlated upwelling hyporheic water DO concentration with ETF survival from the Newmills Burn, northeast Scotland. Like the River Stiffkey the Newmills Burn is a modified stream in an agricultural catchment with high catchment-derived sediment loading. *S. trutta* eggs were installed into artificial redds cut in natural gravels. Malcolm et al. (2003) observed that during high water flows associated with storm events the composition of water in interstitial gravels was dominated by downwelling of water from the stream. However, as water levels receded the dominant flux of water reversed as groundwater upwelling became the dominant component of interstitial water again. Water movement between the stream and interstitial gravels caused the observed variation in DO concentrations within the incubation substrate. Further, Malcolm et al. (2003) concluded that river modification (straightening and deepening) altered the naturally established groundwater-surface water interactions with concomitant implications for *S. trutta* recruitment. This is a probable explanation for poor ETF survival observed on site 2009A in the River Stiffkey, a site that did not accrue much sediment (<1 mm) over the incubation period (2.2%). Given the importance of groundwater upwelling in chalk streams for embryo development, it seems advisable that rehabilitation gravel should be installed into reaches of established groundwater upwelling. Otherwise a sustainable high stream velocity should be associated with rehabilitation gravel to ensure sufficient downwelling into interstitial gravels as long as suspended sediment transport rates remain low.

### 5.9.4 Incubation sediment structure and permeability: comparison between rehabilitation and natural gravels

Whether suspended sediment is excluded, trapped or accumulated within embryo incubation substrate is determined by the ratio between interstitial pore size and suspended sediment size (Frostick et al., 1984; Lisle, 1989). Moreover, this ratio determines whether particles settles in surface sediments or accrue within deeper substrate (Lisle, 1989). The greater the size difference the more susceptible incubation gravels will be to fine sediment accrual within deeper substrate. Well sorted spawning gravels in an environment with high suspended sediment loads are particularly vulnerable to deposition. These sediments will typically accumulate bottom-up (Figure 5.19 and 5.20), reducing permeability and consequently intragravel velocity (Greig et al., 2007). During the redd cutting process the sorting coefficient, an indicator of interstitial void space (Ingendahl, 2001), decreases whilst grain-size $D_{50}$ increases as the fine sediment fraction is removed (Kondolf et al., 1993) and interstitial voids increase. This results in an initial accrual of fine sediment in deep laying substrate, and the coarser the subsurface gravel the deeper and more sustained the intrusion of sediment becomes (Greig et al., 2007). The inverse was observed in Beschta and Jackson's (1979) flume experiments; fine sand ($D_{50}$=0.5 mm) settled within the upper 10 cm of streambed substrate ($D_{50}$=15 mm), in a similar manner to natural gravels observed in this study. It is likely that the greater interstitial voids formed by a high composition of gravels (64>$D$≥16 mm) and a low abundance of coarse sand (2>$D$≥1 mm) within deeper substrate in rehabilitation gravel formed greater median particle diameters ($D_{50}$=22.6 mm and 20.3 mm for 2003 and 2009 respectively) thus enabling greater settling depths.

Sediment composition of redd substrate between the different gravel treatments was not similar prior to the introduction of eyed embryos due to inherent sedimentological differences between rehabilitation and natural gravels. This resulted in a difference in the manner in which sediment <1 mm accrued within redds on different gravel treatments. Rehabilitation gravel was well sorted throughout the deposit and fine sediment (<1 mm) characteristically accumulated bottom-up. The large interstitial pore spaces were readily available for fine sediment (<1 mm) to accumulate and as such reduced permeability for incubating embryos, particularly in the 2003 rehabilitation gravels. The highly variable character and gravel-sandy mix observed in natural incubation gravels in the River Stiffkey reduced interstitial pore to sediment size ratio differences and, as a consequence, deposition occurred mostly in surface substrate (Lisle, 1989). In this manner surface pore spaces were reduced as successively smaller sized sediment particles deposited out of suspension and accumulated. The surface armouring that developed in natural gravels in the River Stiffkey prevented accrual of finer sediments into deeper lying substrate and maintained an area of intragravel permeability beneath surface layers. A similar characteristic has been reported by several authors (Frostick et al., 1984; Lisle, 1989; Greig, et al., 2007). However, because pre-incubation substrate had a well sorted sediment structure, consistent with the natural redd cutting process (Kondolf et al., 1993), natural gravels underwent an initial accrual of sediment (<1 mm) in deep laying substrate as coarse sand preferentially settled within the surface layers. Sands (2>D≥1 mm), assumed transported as bedload, were characteristically deposited within the surface 10 cm of redds cut on natural control sites. Similar deposition was not observed in rehabilitation gravel redds.

Due to their unique gravel composition, structure and catchment geology, a surface armour is a characteristic feature of chalk streams (Crisp, 1993). Whilst surface substrate with a high composition of sand might inhibit alevin emergence (Kondolf, 2000; Hartman and Hakala, 2006), Crisp (1993) indicated that *S. trutta* can emerge through up to 8 cm of sand within incubation substrate. Surface armouring of sediment observed in natural embryo incubation substrate in the River Stiffkey should therefore not inhibit alevin emergence. In their laboratory experiments with *S. salar* eggs, Lapointe et al. (2004) observed a significant negative correlation between ETF survival and intrusion of fine grained sediment (<0.063 mm) into incubation substrate ($D_{50}$ = 26 mm). However, the authors noted that ETF survival can be improved at greater interstitial gravel velocities with a greater contribution of sand (2>$D$>0.63 mm) to the incubation substrate and a low abundance of fine grained sediments (<0.063 mm) (Lapointe et al., 2004).

Once Whey Curd, a natural gravel site dominated by fine sediment (<1 mm) input, had been excluded from the regression analysis, a positive correlation was observed between ETF survival and redd velocities at the natural gravels sites Water Hall and Fort (Spearman's Rank-Order, $r_s$=0.544, $p$<0.05). Streambeds of North Norfolk streams typically consist of a sandy matrix overlain by shallow gravel deposits (Milan et al., 2000). Although these streams exceeded the fine sediment (<1 mm) threshold for 50% ETF emergence (14%), Milan et al. (2000) maintained that the combination of a high abundance of medium to coarse sand (2>$D$>0.125 mm) and low fine sediment (<0.063 mm) composition were key factors that retained interstitial permeability within incubation substrate. This may explain the relatively high ETF survival observed in both the Water Hall and Fort sites within the natural gravel treatment as each site had elevated compositions of fine sediment (<1 mm). However, the markedly high composition (>60%) of sediments <1 mm within embryo incubation sediments at Whey Curd exceeded a threshold level of the ratio between interstitial void size to sediment size and choked the incubation gravels, and as such had a correspondingly very poor mean ETF survival. Clearly excessive sediment input can override the association between redd velocity and ETF survival and is an indication of the complex dynamic between velocity and sediment (<1 mm).

### 5.9.5 Spatial distribution of ETF survival linked to sediment sources in the River Stiffkey

Sedimentation is a naturally occurring stream process. However, land-use and poor land management has increased sedimentation rates, exacerbating the associated detrimental ecological implications (Wood and Armitage, 1997). Winter rainstorm events erode and transport sediment from the arable catchment to the River Stiffkey channel using farm tracks and rural roads as conduits (Figure 3.11, Chapter 3). Walling and Amos (1999) and Walling et al. (2006) described a similar process of sediment delivery in chalk stream catchments of the Rivers Piddle (Dorset), Pang and Lambourn (Berkshire).

Sediment is transported downstream in a series of pulses at decadal timescales (Syvitski, 2003). The magnitude of sediment pulsing or migration through the river channel decreases with distance downstream (Montgomery and Bolton, 2003). However, sediment transport increases in those systems where artificially created flood banks isolate a river from its floodplain (Knighton, 1984; Montgomery and Bolton, 2003). Much of the mid to lower River Stiffkey has flood banks which are disconnected from its floodplain and as such it is anticipated that sediment storage within the channel has been artificially increased. Walling et al. (2006) noted that very little (1%) of fine sediment eroded from within the catchment was discharged into the sea in the Pang and Lambourn catchments (Berkshire, UK). In their investigation of sediment dynamics in the upper River Piddle in Dorset, Walling and Amos (1999) found that sediment was slowly mobilised downstream during summer months. A similar sediment dynamic was observed in the River Stiffkey. The Wighton village road bridge, between sites Water Hall and 2003A (Figure 5.21), was a major point source of catchment-derived sediment input during rainfall events where large volumes of sediment were deposited in the river channel (see Section 3.2.2, Figure 3.11, Chapter 3).

Downstream sediment dispersal was reflected in the marked decline in ETF survival at rehabilitation gravel site 2003A, and the gradual increase in ETF survival as the contribution of this sediment decreased on successive sites (Figure 5.21). All 2003 sites had low but comparable ETF survival rates, suggesting a common limiting factor. There was a significant increase in ETF survival between the most spatially separated rehabilitation sites, 2003A and 2009J (Figure 5.21a and b; Mann-Whitney, $p<0.05$, Table 5.2), further suggesting a determining mechanism controlling ETF survival over this spatial range. Furrows cut through the width of the buffer strip connecting the adjacent arable field and river channel at the Whey Curd gravel site was identified as a further sediment input source (Figure 5.16 and 5.21). During rainstorm events these furrows facilitated field drainage funnelling excessive sediment directly into the river. Grain-size distribution analysis indicated a high contribution of sediments $D<1$ mm in the incubation gravel, as well as an elevated level of sediment $D<2$ mm intrusion during embryo incubation at Whey Curd (Figure 5.16). ETF survival was also poor at this site (4.5%), being similar to ETF survival observed on the 2003 rehabilitation spawning gravels downstream of the Wighton village bridge (Figure 5.21b, Table 5.1). The spatial proximity of the 2003 rehabilitation gravels, as well as natural gravels at the Whey Curd site, to high inputs of catchment-derived sediment played a key role in the deterioration of ETF survival at these sites. The introduction of catchment-derived sediment into a gravel-dominated incubation substrate altered the framework-matrix composition, blocked substrate interstices, reduced interstitial velocity and DO concentrations (Petts, 1984; Olsson and Persson, 1986; Chapman, 1988; Petts, 1988; Graham, 1990, Kondolf et al., 1993, Sear, 1993).

Evidence for agriculturally-derived sediment inputs is compelling (see section 3.2.2, Chapter 3) and has been recognised by the UK government as a major threat to salmonid stock decline in England and Wales for almost 2 decades (see Theurer et al., 1998). The Environment Agency (EA) in collaboration with the Soil Survey and Land Research Centre (SSLRC) led an evidence based investigation to determine whether catchment-derived sediment affected salmonid stocks, specifically through deterioration of spawning habitat, in England and Wales (Theurer et al., 1998). This study concluded that sediment pollution was indeed widespread, particularly in rural areas, and a significant cause of salmonid decline. Although fine sediment (<0.125 mm), derived from catchment surfaces, comprised a significant element of suspended sediment loads (Walling et al., 2003), it is likely that the sand-sized particle (>0.125 mm) load had been eroded from the channel bed and bank (Walling et al., 2000).

### 5.9.6 The temporal scale of ETF survival: rain, discharge and sediment deposition

Although sediment intrusion into incubation substrate has the potential to control salmonid populations, the negative relationship between fine grained sediment (<1 mm) and ETF survival is dependent on site specific as well as catchment processes operating at variable time scales. Annual and seasonal variation in water level can have a considerable impact on this relationship (Hartman and Hakala, 2006). The increase in ETF survival during 2012, apparent for both the 2009 rehabilitation and natural gravels in the River Stiffkey, was associated with climatic variation and subsequently stream discharge. The River Stiffkey catchment has glacial deposits in the upper reaches (see section 3.2.1, Chapter 3) that decrease the typically moderated chalk stream response time to rainfall. During rainstorm events sediment was eroded and transported from the catchment to the river channel, as observed during convective rainstorm events (Figure 3.11; Chapter 3). A higher stream discharge during the 2011 study period was associated with significantly lower ETF survival than observed during the reduced discharge levels of the 2012 study (Mann-Whitney, $p<0.05$, Table 5.2). Reduced rainfall during the winter months of early 2012 (Figure 5.1) potentially eroded less sediment from the arable catchment relative to 2011 and thus revealed how significantly detrimental catchment-derived sediment was to *S. trutta* ETF survival.

High flow events are associated with an increase in suspended sediment deposition into streambed sediment (Lisle, 1989; Wood and Armitage, 1997; Ingendahl, 2001). This relationship had been noted elsewhere. For example, Greig et al. (2005a) observed greater sediment deposition during higher flow events on the Rivers Aran and Blackwater resulting in a significant correlation between the accumulation of sediment (<1 mm) and reduced interstitial velocities (Greig et al., 2005a). Acornley and Sear (1999) measured fine sediment deposition into *S. trutta* embryo incubation gravels on the River Test, Hampshire and reported a seasonal pattern of increased deposition during the high discharge periods of winter and early spring. Further, these authors observed that 96% of the annual suspended sediment load was transported during the *S. trutta* spawning and incubation period. Based on calculated deposition rates, they determined that it would take just 25 days for the fine sediment fraction removed during the redd cutting process to re-accumulate into the incubation substrate. Additionally, Zimmermann and Lapointe (2005) described a reduction of interstitial velocities in response to rain storm events that transported sediment from land surfaces into the river Cascapédia River channel, Canada. This study suggest that sediment loading of the River

Stiffkey is dependent on catchment processes, such as land-use, but driven by climatic variables that operate at both the seasonal and inter-annual scale.

### 5.9.7 Longevity of rehabilitation gravels as a suitable *S. trutta* spawning habitat: temporal and spatial considerations

Morphosedimentary succession of rehabilitation gravel (see section 4.4, Chapter 4) was associated with an observed temporal degradation of ETF survival in the River Stiffkey. Rehabilitation gravels installed in 2003 displayed poor mean ETF survival over both study periods, as well as lower mean water velocities with narrower distributions than those gravels installed in 2009. Further, the older 2003 rehabilitation gravel treatment did not have a positive ETF survival response to the reduced sediment load pressures during the 2012 study as observed on the 2009 rehabilitation and natural gravels. The 2003 rehabilitation gravels were characterised by a sedimentary structure that reflected a greater length of time exposed to stream conditions than rehabilitation gravel installed in 2009; greater levels of fine sediment (<1 mm) ingress, remobilisation of the more mobile gravel size fraction ($30>D_{50}\geq16$ mm) and gravel ($64>D\geq16$ mm). In addition, subsidence has exposed cobbles ≥64 mm used to anchor the instalments onto the streambed. Natural gravel sites exhibited a similar silt/sand gradient as observed in the 2003 rehabilitation gravel, although sediment grain-size variation was greater. The natural gravel treatment was characterised by high variability in sediment grain-size as well as redd velocity and had an associated good ETF survival response (Figure 5.7). As silt and sand accrued over time within rehabilitation gravel it became quasi-naturalised. However, due mainly to the larger gravel size-range and poor variation in stream velocity, rehabilitation gravel will reach a stable state of lower variation in sediment grain-size relative to natural gravels. Although the more recently installed 2009 rehabilitation gravels had a greater ETF survival, it is very probable that sedimentary composition and structure will change over time as both rehabilitation gravel treatments were constructed to similar specifications. The dominance of gravels ($64>D\geq16$ mm) within the incubation substrate create large interstitial spaces available for fine sediment ($D<1$ mm and $2>D\geq1$ mm) intrusion from the large supply of readily available sediment, and redistribution of surface gravel ($30>D\geq16$ mm) over time will be associated with a decline in ETF survival.

There are few comparable studies on rehabilitation gravel, however, the morphosedimentary succession and associated temporal degradation of ETF survival results of this study were similar to those reported by Pulg et al. (2013) on rehabilitation gravel in the Moosach River, a chalk stream in southern Germany. The Moosach River catchment also has a high rate of sediment erosion and mobilisation. Pulg et al. (2013) monitored ETF survival between 2004-2008 on two types of rehabilitation gravel features; cleaned natural gravels, and gravel installed in a pool-riffle formation. Gravels were installed in the size range (32>*D*>16 mm) with <1% sediment <1 mm. 190 fertilised *S. trutta* eggs installed in each mesh egg-box, and intern egg-boxes were inserted into rehabilitation and control sites. Freeze cores of post-incubation substrate, interstitial DO and fish surveys by means of mark and recapture electric fishing methods were also conducted. ETF survival was highly variable (0-93%) with a mean of 46%. Pulg et al. (2013) observed a negative correlation between ETF survival and percentage fine sediment (<0.85 mm) and a positive correlation between ETF survival and interstitial DO concentration. Due to a significant increase in percentage fine sediment (<0.85 mm) over the 4 year period (from 0 to 10% contribution) the authors noted a $D_{50}$ reduction in rehabilitation gravel from 22 mm to 13 mm. Gravel treatments began to resemble a similar sediment composition after 1 year. ETF survival declined over the study period suggesting morphosedimentary succession of the rehabilitation gravel; years 1-2 had >50% ETF survival and <50% in years 3-4. The authors concluded that rehabilitation gavels in the Moosach River would degrade to a state considered wholly unsuitable for embryo development within 6 years. Similar to the Moosach River, in the River Stiffkey 8 years of accumulated fine sediment derived from land-use sources within the catchment have inhibited gravel interstitial space, reduced permeability and rendered the 2003 rehabilitation gravels sites inadequate for *S. trutta* recruitment.

### 5.10 Conclusions

Rehabilitation gravel was installed into the River Stiffkey as part of a river conservation project seeking to address a purported lack of suitable natural spawning habitat and thereby augment *S. trutta* recruitment. Excessive catchment-derived sediment inputs have prevented rehabilitation gravel from adequately achieving the intended objective. This study suggests that accumulation of excessive fine grained sediment has altered the structure and composition of rehabilitation gravel, decreasing gravel permeability, interstitial velocities and DO with detrimental impacts on embryo survival. The *S. trutta* population in the River Stiffkey is likely controlled by spatial and temporal scales of sediment input, resulting in a vulnerable *S. trutta* population reliant on a small number of spawning gravel sites for recruitment.

Rehabilitation gravel, installed to remedy the problem, undergo a physical morphosedimentary succession, driven by a large catchment-derived sediment supply. The proximity of rehabilitation gravel to sources of sediment input increased the susceptibility to fine sediment deposition and subsequently degradation of spawning quality. Embryo survival, however, increased with distance downstream over successive sites from the point of sediment input. In this manner, sediment deposition effected a compositional change in the incubation substrate of rehabilitation gravel resulting in a morphological succession, and an associated decline in ETF survival, over the short-term. However, it is likely that the sediment loading of the channel is sufficient such that the primary driver for morphological succession is temporal rather than spatial based on the sediment transfer dynamics that operate at variable time scales within the catchment. Seasonal and annual variation in precipitation and land-use management (sediment availability) can have a considerable impact on this relationship. Embedded within this is the spatial relationship between the location of rehabilitation gravel sites and points of sediment input that have a direct impact on embryo survival.

Rehabilitation gravel that mimics an upland stream-type spawning gravel composition are not suited to lowland chalk streams and their characteristic contemporary high sediment loads and low velocity regimes. This is largely due to the minor sand mode present in such rehabilitation gravels. Although the River Stiffkey has high sediment (<1 mm) loads, the sediment structure and composition of natural gravels maintains interstitial permeability and thereby relatively high ETF survival. Rehabilitation gravel should therefore look to better mimic natural gravel structure for greater sustainability on a local scale. Pasternack et al. (2004) and Barlaup et al. (2008) argued that rehabilitation gravel should be composed of a more heterogeneous mix of sediment sizes to prevent redistribution of the smaller mobile gravel sizes, and to reduce finer grained sediment accrual. Further, a wide range of gravel sizes will support a wider size range of spawning fish as salmonid size and spawning gravel grain-size are directly correlated (Kondolf and Wolman, 1993; Kondolf, 2000; Armstrong et al., 2003; Louhi et al., 2008).

Rehabilitation of catchment scale processes, particularly the large sediment supply and low hydraulic regime, over variable spatial and temporal scales are important management considerations to sustain spawning habitats and increase population recruitment. Embryo survival can be significantly improved through the control of sediment supply at source (Scott and Beaumont, 1994; Cefas, 1999; Greig et al., 2005a; Pulg et al., 2013), which will also increase rehabilitation gravel longevity. Therefore a management plan aimed specifically at reducing the levels of sediment entering the river should have a direct impact on population recruitment in the River Stiffkey over the medium to long term. The 2009 rehabilitation gravels as well as the few remaining natural gravel sites could respond well to a reduction in sediment input, however, the 2003 rehabilitation gravels may now be unsuitable for embryo development. Whilst natural and 2009 rehabilitation gravels contain an abundance of suitable spawning gravel (64>$D$≥16 mm), the loss of surface gravel 30>$D_{50}$≥16 mm and exposure of the anchoring cobbles ≥64 mm has rendered the 2003 rehabilitation gravels wholly unsuitable for *S. trutta* spawning.

An effective management strategy that addresses the reduction of sediment input at the catchment scale (based on land management best practice) and ensures greater variability in stream velocity at the macrohabitat scale is required to prevent the 2009 rehabilitation gravels from deteriorating into a similar unrecoverable state. Catchment scale fine sediment management can be achieved through a systematic desk-study based approach that delineates topographic key flow pathways and identifies key sources of sediment input. The application of each key flow pathway to sediment conveyance requires ground truthing. Walkover surveys can further identify poor land drainage practice and additional sources of sediment input. Further introduction of rehabilitation gravel should consider the relationship between sedimentation and velocity, for example locating gravel in areas of natural scour based on the principles of hydrogeomorphology. Additionally, management strategies should consider the value of continued monitoring at a greater temporal and spatial scale in accordance with the initial objectives. Currently there is a lack of available data contributing to existing knowledge of rehabilitation gravel function and further research is needed to improve management guidance. Future studies would benefit from sampling pre-incubation sediment composition from each natural gravel treatment site, preferably each redd, in order to capture the high natural sediment grain-size variability between sites. Post-embryo incubation sediment analysis at this scale would enable greater understanding of the effect sediment structure and composition have on ETF survival response to the change in redd sediment composition over the embryo incubation period.

# 6 A spatial analysis of juvenile *S. trutta* life-stage specific habitat: the implications for population recruitment

## 6.1 Introduction

Habitat requirements of *S. trutta* are complex and life cycle dependent (Heggenes et al., 1999; Armstrong et al., 2003). Access to suitable habitat once fry have emerged from spawning gravels is critical to population recruitment (Heggenes et al., 1999). The spatial proximity between key habitat types required by juveniles during their first year is important; it determines accessibility based on the capacity of fry to migrate between life-stage specific habitat types. *S. trutta* production can therefore be delineated, and the physical scale for the enhancement of population recruitment through rehabilitation schemes can be defined, by the natural habitat attributes throughout the river channel (Kocik and Ferreri, 1997).

As a spawning habitat, rehabilitation gravels play a key role in defining *S. trutta* recruitment in the early life-stages. Survival of juvenile fish, once emerged from rehabilitation gravel, is dependent on accessing suitable habitat within a maximum migration capacity associated with that life-stage (Elliott, 1981; Ottaway and Clarke, 1982; Elliott, 1987; Klemetsen et al., 2003). Low abundance of, or spatially fragmented, juvenile specific habitat reduces *S. trutta* recruitment, an aspect of population management often referred to as a pinch-point or a bottleneck (Bohlin, 1977; Egglishaw and Shackley, 1977; Elliott, 1989; Armstrong and Griffiths, 2001). Installation of rehabilitation gravel conducted in spatial isolation from other key juvenile specific habitat types may transfer a recruitment bottleneck from embryo development and fry emerging (spawning gravel) to other early life-life-stages. Improvements to population recruitment through the installation of rehabilitation gravel alone might not be successful should other life-stage habitat types be spatially fragmented or scarce.

An assessment of functional habitat units (FHU), river reaches suitable for *S. trutta* production based on the spatial proximity between life-stage specific habitat constrained by the maximum migration capacity to move between these habitat types, was determined in this chapter consistent with Kocik and Ferreri (1997). The area ($m^2$) of specific habitat over a length of river channel where these habitat types occur within the migration ability of juvenile *S. trutta* define areas of juvenile production. Lengths of river where FHU are absent, due either to spatial fragmentation or poor habitat abundance, are invaluable for river mangers seeking to improve *S. trutta* population recruitment. A walkover habitat survey, <1 km upstream of Wighton village to the confluence with the Binham Stream (<1 km downstream of Warham), quantified juvenile habitat types and was conducted between 6-9 July 2011 (see section 2.3.4, Chapter 2). Specific habitat types surveyed included: rehabilitation gravel, marginal habitat, undercut banks, overwintering refugia, high flow fry refugia, large woody debris, vegetation stands and overhanging vegetation. Discrete identification of natural spawning gravel habitat was not conducted due to the high spatial variability of the streambed sediment composition. Given such variability, the gravel streambed habitat, a matrix-filed coarse gravel substrate that included spatially variable natural spawning habitat, was surveyed. Each habitat type was associated with an early life-stage of *S. trutta*.

After emergence from spawning gravels, fry (yolk-sac recently resorbed, exogenously feeding) require access to nursery habitat, typically within the channel margins, for early development over the first summer (Elliott, 1987; Klemetsen et al., 2003). Fry are susceptible to increased stream velocities >0.02 m $s^{-1}$ (Armstrong et al., 2003; Hendry et al., 2003) and as such migration distances of 10 m and 40 m were used in the analysis of FHU to account for the difference between low and high flows respectively (Elliott, 1981; Ottaway and Clarke, 1982; Elliott, 1987). As fry grow over the summer months they develop into parr, and although they do not move far from their natal spawning gravels, they seek out alternative rearing habitat with velocity tolerances ≤0.70 m $s^{-1}$ (undercut banks, large woody debris, vegetation stands and overhanging vegetation) (Heggenes et al., 1999; Armstrong et al., 2003; Hendry et al., 2003). The migration from rearing habitat to establish overwintering refuge is of significance for population recruitment as it is linked to juvenile survival (Solomon and Templeton, 1976; Brown et al., 2001). A migration distance of 100 m from spawning gravels to overwintering habitat, consistent with Solomon and Templeton (1976) and Brown et al. (2001), was used for FHU analysis.

Due to the modified character of the River Stiffkey channel, stream flow was mostly homogenous across the stream width. Alternative flow biotopes were therefore defined by stream depth, associated flow type and substrate size, consistent with Raven et al. (1997) and Padmore (1997): run (<30 cm depth with a rippled flow type over cobble, gravel and sand), glide (30-60 cm, smooth boundary, no eddies, sand and silt), deep glide (60-90 cm, smooth boundary, no eddies, sand and silt), very deep glide (90-120 cm, smooth boundary, no eddies, sand and silt) or pool (>120 cm, scarcely perceptible flow, silt). The habitat survey was digitised in ArcMap (v10.2) and FHU determined, constrained by the maximum juvenile dispersal distances between life-stage specific habitat types. Migration distances were measured downstream from spawning gravels, both rehabilitation and natural sites, as observed in chalk streams elsewhere (Solomon and Templeton, 1976; Moore and Scott, 1988). If maximum migration distance did not include the target life-stage habitat type, that FHU was not considered. Overlapping FHU were aggregated into a single compound FHU, the greater the area ($m^2$) the greater the *S. trutta* production potential (Kocik and Ferreri, 1997). Delineated FHU defined the spatial scale of *S. trutta* production based on physical habitat characteristics. Such analysis is invaluable for river management to develop rehabilitation strategies that can be delivered at the naturally required spatial scale to complement existing habitat features.

The aim key of this chapter was to determine limitations to population recruitment based on the abundance of, and spatial relationships between, juvenile life-stage specific habitat. Defining FHU within the study area provided an indication of favourable locations to install rehabilitation gravel based on the proximity to other life-stage specific habitat types. Specifically, this chapter aims to:

- investigate the spatial relationship between key early life-stage specific habitat types that may limit *S. trutta* population recruitment in the River Stiffkey during the first year and in so doing define areas of juvenile production
- ascertain an appropriate scale of stream management based on the abundance and spatial relationships between life-stage specific habitat
- determine suitable locations to install rehabilitation gravel based on these analyses

### 6.2 Abundance of early life-stage specific habitat and flow biotopes

Early life-stage habitat types in the River Stiffkey had a narrow range of mostly small areas (mean <10 $m^2$) with tight clustering around median values (Figure 6.1, Table 6.1). Abundance of marginal and overhanging vegetation refugia was relatively high, n=343 and n=258 respectively, however, mean area coverage of each was low, 2.7 $m^2$ and 6.9 $m^2$ (Table 6.1). Conversely, rehabilitation and streambed gravel had low abundance, n=12 and n=25 respectively, but greater mean area cover, 196.9 $m^2$ and 264.4 $m^2$. Rehabilitation and streambed gravel habitats had significantly greater median areas than other habitat types (Kruskal-Wallis, $p$<0.05, Table 6.2). Although rehabilitation gravel had a greater mean area (196.9 $m^2$), a small range of values around the median reflected the similar specifications to which they were constructed. The median area of streambed gravels was significantly lower (32.7 $m^2$) than rehabilitation gravel (192.6 $m^2$) (Mann-Whitney, $p$<0.05, Table 6.2), however a greater range of values was observed in stream gravels reflecting the observed natural variability. Naturally available gravels suitable for *S. trutta* spawning occurred as smaller spatially fragmented areas within the streambed habitat type. Discrete identification of these spawning gravels was improbable due to the spatially fragmented nature and overlaying deposits of fine sediment. Modification to the stream reduced channel form morphology and heterogeneity. A discrete pool-riffle morphology did not exist. The streambed gravel habitat type was longitudinally uniform with few apparent bedforms at the macrohabitat scale. The River Stiffkey was characterised by low stream flows, with little variation within either the longitudinal or horizontal (length and width) dimension. The persistence of low flows provided a good indication of the difficulty in locating discrete natural spawning gravel habitat. Areas of stream channel described as glide biotopes (glide, deep glide, very deep glide) dominated stream flow (Figure 6.2, Table 6.3), and was likely a response to the modified character of the stream channel. The very deep glide biotope had a significantly greater median area (169.5 $m^2$) than the deep glide (92.1 $m^2$). Although rehabilitation gravel had been installed, the run flow biotope had a low abundance. Moreover, there was no occurrence of a flow biotope of greater magnitude. There was a low abundance of the pool biotope, n=8 (Table 6.3) with a significantly lower median area coverage (40.1 $m^2$) than any other flow biotope (Mann-Whitney, $p<0.05$, Table 6.2).

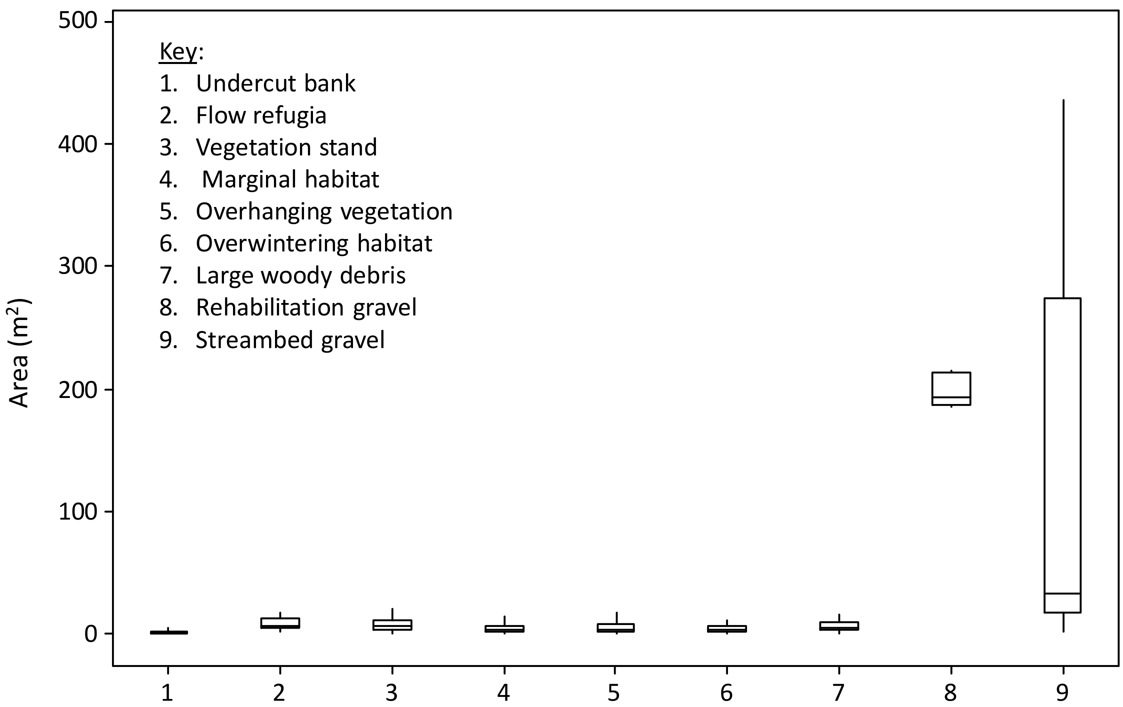



**Figure 6.1** Summary boxplot of life-stage specific habitat type area ($m^2$). Outliers have been removed to expand the *y*-scale of the main data. Discrete habitat areas within each habitat type were mostly small, apart from streambed gravels (9).

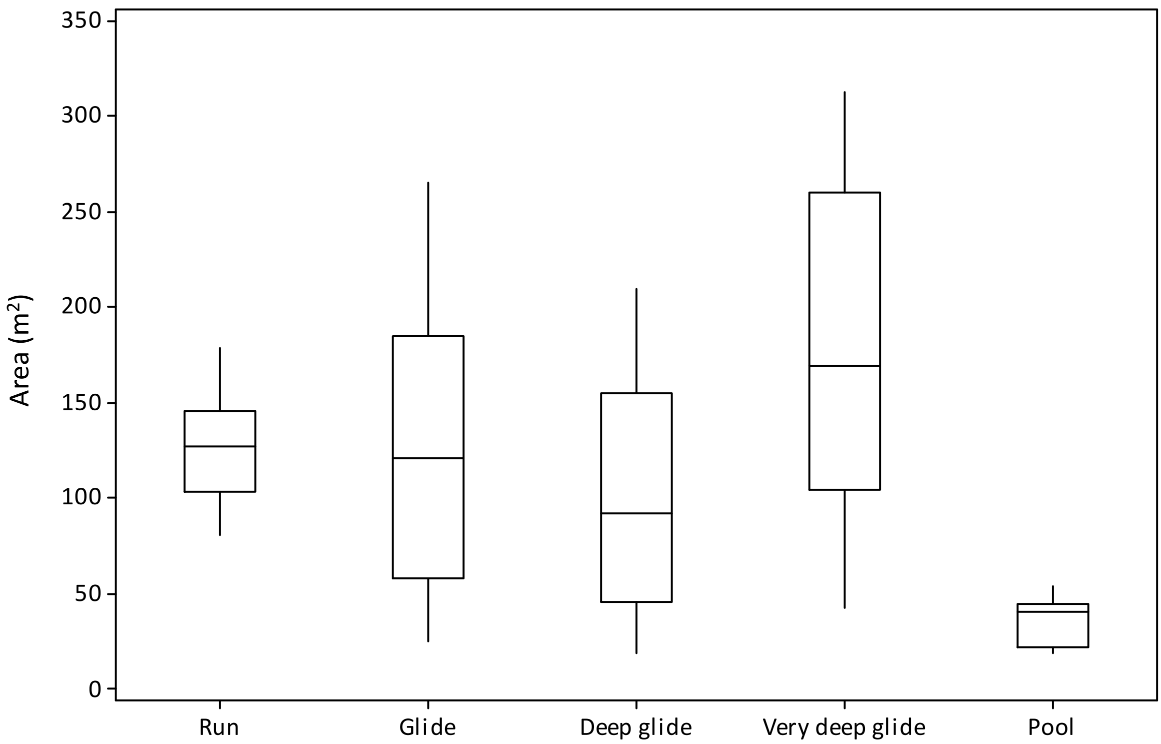



**Figure 6.2** Summary boxplot of area ($m^2$) of stream flow biotopes. Outliers have been removed to expand the *y*-scale of the main data. Stream flow area ($m^2$) was dominated by low flow biotopes.

**Table 6.1** Summary of the mean (± S.D) area ($m^2$) of early life-stage specific habitat types observed in the study site. Abundance and area ($m^2$) were mostly low. Marginal and overhanging vegetation were, however, observed in relatively greater abundance and area ($m^2$).

| | | **Area ($m^2$)** | | |
|---|---|---|---|---|
| **Habitat type** | **N** | **mean** | **±** | **SD** |
| Undercut bank | 7 | 1.7 | ± | 1.4 |
| Flow refugia | 5 | 8.4 | ± | 5.6 |
| Vegetation stand | 13 | 8.8 | ± | 8.3 |
| Marginal | 343 | 2.7 | ± | 4.9 |
| Overhanging vegetation | 258 | 6.9 | ± | 8.6 |
| Overwintering | 45 | 4.9 | ± | 4.8 |
| Large woody debris | 22 | 8.8 | ± | 10.8 |
| Rehabilitation gravel | 12 | 196.9 | ± | 40.1 |
| Streambed gravel | 25 | 264.4 | ± | 616.1 |

**Table 6.2** Summary results of Kruskal-Wallis and Mann-Whitney U analysis for difference between habitat area ($m^2$) and stream flow biotope.

| | | | **Mann-Whitney U** | | | | |
|---|---|---|---|---|---|---|---|
| | | **Kruskal-Wallis** | **streambed** | **run** | **glide** | **deep glide** | **very deep glide** |
| **Habitat** | **Type** | 1 | - | - | - | - | - |
| | **rehabilitation** | - | 1 | - | - | - | - |
| **Flow** | **Biotope** | 1 | - | - | - | - | - |
| | **glide** | - | - | 0 | - | - | - |
| | **deep glide** | - | - | 0 | 0 | - | - |
| | **very deep glide** | - | - | 0 | 0 | 1 | - |
| | **pool** | - | - | 1 | 1 | 1 | 1 |

**Table 6.3** Summary of mean (± S.D) area ($m^2$) flow biotope. Glide biotopes dominated stream flow. A riffle-pool morphology was absent from the study site.

| | | **Area ($m^2$)** | | |
|---|---|---|---|---|
| **Flow biotope** | **N** | **mean** | **±** | **SD** |
| Run | 15 | 142.8 | ± | 98.2 |
| Glide | 27 | 409.6 | ± | 867.0 |
| Deep Glide | 27 | 121.4 | ± | 132.4 |
| Very Deep Glide | 14 | 247.4 | ± | 248.8 |
| Pool | 8 | 36.2 | ± | 12.6 |

## 6.3 Physical constrains to rehabilitation

The River Stiffkey had few barriers to *S. trutta* migration. Two potential barriers to free passage were observed, however neither of these were formalised weir structures, consisting of loose gravels, cobbles and cement bricks lined together to form weir-like structures (Figure 6.3b and c). The first barrier, surveyed in the upper reaches of the study site (TF 94142 39109), had a head difference of approximately 20 cm and was used to impound water for an abstraction uptake hose. Low water levels may inhibit *S. trutta* passage over this structure. Fine sediment deposition dominated the upstream reach whilst coarser sized substrate were observed downstream. The second structure, next to the Binham Road bridge in Warham (TF 95090 41649), had an estimated head difference of approximately 10 cm and impounded water to act as a cattle drink. Upstream abundance of fine sediment was deposited out of suspension, which dominated the composition of surface sediments there. The downstream area had been reinforced with gravel sized sediments to enable cattle to cross. Greater water flow here spilling over the barrier maintained a composition free of fine sediment. Stream depths at low flows would likely limit *S. trutta* passage over the gravel apron downstream of the weir. High water levels would potentially drown out the structure. Neither weir structure had a plunge pool to assist fish passage.

Cattle access to the river channel had mostly been fenced off though the Holkham Estate stretch of the study site (from TF 94440 40032 downstream of Wighton village to TF 95577 41922 at the confluence with the Binham Stream; Figure 2.1, Chapter 2) as part of the LNS project strategy for the River Stiffkey. Damage to the banks was obvious in all areas were cattle previously had access to the stream (Figure 6.3a and d). Fine sediment deposition dominated surface substrate composition in these reaches. Two areas with free access to the river for cattle were observed: one slightly upstream of the Binham Road bridge (alongside the weir, Figure 6.3c), and the other slightly downstream of an Iron Age fort midway through the study site. Poorly maintained fencing however in the upper reaches of the study site, upstream of Wighton village, did not restrict cattle access. Banks in these areas were not stable and were a potential source of abundant sediment input.

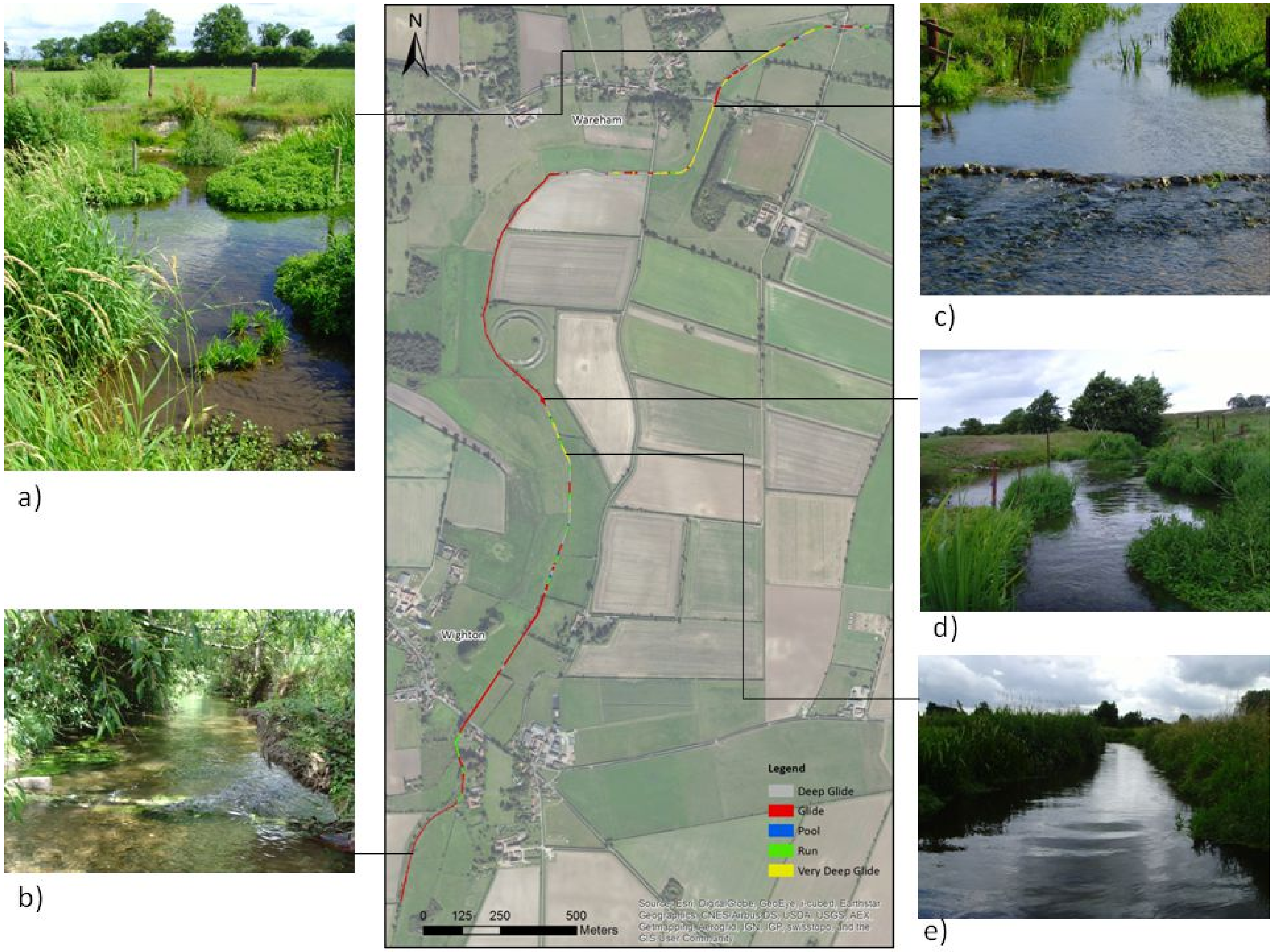



**Figure 6.3** Location of cattle access and associated river bank damage (a) and (d), two weirs (b) and (c), as well as stream flow distribution throughout the study site and characteristic flow biotope (e). The study site was characterised by low stream flow and fine sediment deposition was associated with channel modification.

### 6.4 Wooded and meadow stream reaches: implications of dredging on flow biotopes

Two distinct stream reaches based on stream flow biotope and habitat types were apparent: a wooded reach characterised by well vegetated stream banks and typically shallow stream depths; as well as an open meadow-like reach characteristically treeless but with vegetated marginal buffer strips and greater stream depths (Figures 6.4 and 6.5a-f). Hereafter these stream reach types are referred to as *wooded* and *meadow* reaches. Two of each stream reach type were observed in an alternating fashion: wooded reaches occurred at 0-755 m and 1865-2635 m whilst meadow reaches were observed between 755-1865 m and 2635-4050 m measured from the upstream most point of the study site (Figure 6.4). Wooded reaches had a greater range of stream widths due to vegetative encroachment and greater physical heterogeneity (Figure 6.5a-c). Shallower stream depths were more amenable to and contained a greater abundance of primary production (macrophyte growth). Streambed substrata were also less well sorted and contributed more to bed roughness with a greater degree of armouring. Meadow reaches had a greater level of historical management, predominantly dredging, that had removed much of the natural streambed substrate (Figure 6.5d-f). Dredging was not as evident, if at all, throughout the wooded reaches.

Stream flow was dominated (>80%) by the various glide biotopes (glides, deep glides and very deep glides) (Figures 6.3 and 6.6). 55.5% of the total cumulative flow area ($m^2$) was a glide biotope (Figure 6.6). Glides biotopes were characterised by a smooth consistent low flow condition with little surface undulation differentiated from each other by stream depth, ranging between 30 cm (glide) to 120 cm (very deep glide). 17% of the surveyed river section was described as a very deep glide, 90-120 cm depth, with very little (11%) of the total flow biotope faster than a glide (Figure 6.6). The key substrate characteristics associated with deeper flow biotopes (glides, deep glides, very deep glides and pools) were fine grained sediments; the deeper the biotope the greater the deposition of suspended sediment (Table 6.4).

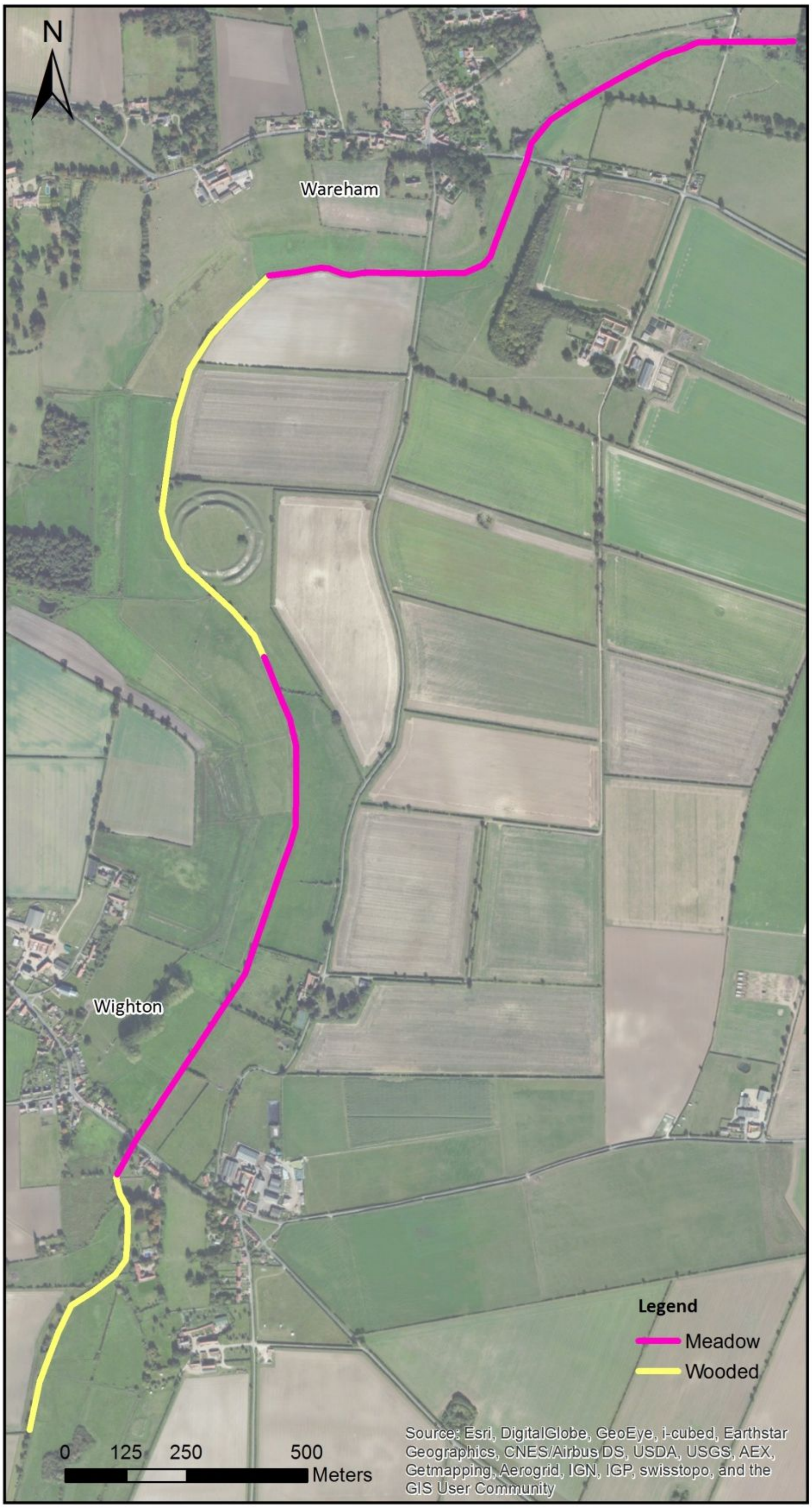



**Figure 6.4** Location of wooded and meadow reaches throughout the study site. Over-deepened meadow reaches were a dominant feature and the location of rehabilitation gravel. Shallow water and vegetated stream banks characterised wooded reaches.

Wooded reaches were characterised by a glide biotope (30-60 cm depth); 52.2% of the total glide biotope was observed in wooded reaches (Table 6.5), whilst <90% of flow in wooded reaches was a glide biotope (Table 6.6). However, run biotopes contributed <23% of the total contribution of flow in the upstream-most wooded reach (Table 6.6), and just 3.5% of the total observed run biotope area ($m^2$) occurred in the upstream-most wooded reach (Table 6.5). The downstream-most wooded reach was described as 100% glide biotope. Key substrate characteristics of wooded reaches were dominated by medium gravel to sand with spatially fragmented areas of coarser gravel, associated with run biotopes, in the upstream most wooded reach only. Meadow reaches had relatively lower flow biotopes than observed in wooded reaches, and by association a greater composition of finer grained sediment stored within streambed substrate. Dredging in meadow reaches impacted flow and increased suspended sediment deposition. Meadow reaches were characterised by deep flow biotopes (glides, deep glides, very deep glides and pools). 100% of the deep glide, very deep glide and pool biotopes, which were not observed in wooded reaches, occurred in meadow reaches (Tables 6.5 and 6.6). The downstream meadow reach had a greater contribution to the total flow area ($m^2$) of deep glide and very deep glide biotopes than the upstream meadow reach, 61.8% and 79.1% respectively.

Just 1% of cumulative area of the surveyed channel was described as pool habitat (>120 cm depth) (Figure 6.6), which all occurred in meadow reaches (Tables 6.5 and 6.6). However, 60.9% of the pool biotope occurred in the upstream meadow reach and was largely associated with rehabilitation gravel. Further, rehabilitation gravel structures increased water flow very locally from a long glide to a series of shorter runs (<30 cm stream depth). In this manner, rehabilitation gravel increased flow biotope diversity in meadow reaches; the run and pool biotope contributed 10.4% and 2.2% respectively to the total flow area ($m^2$) each associated with rehabilitation gravel. Rehabilitation gravel mimicked a pool-riffle sequence, however, this was largely due to the location of rehabilitation gravel in the over-deepened meadow reaches. Here long stretches of dredged streambed occurred between each gravel instalment forming glides with the occasional pool at the downstream end of rehabilitation gravel sites.

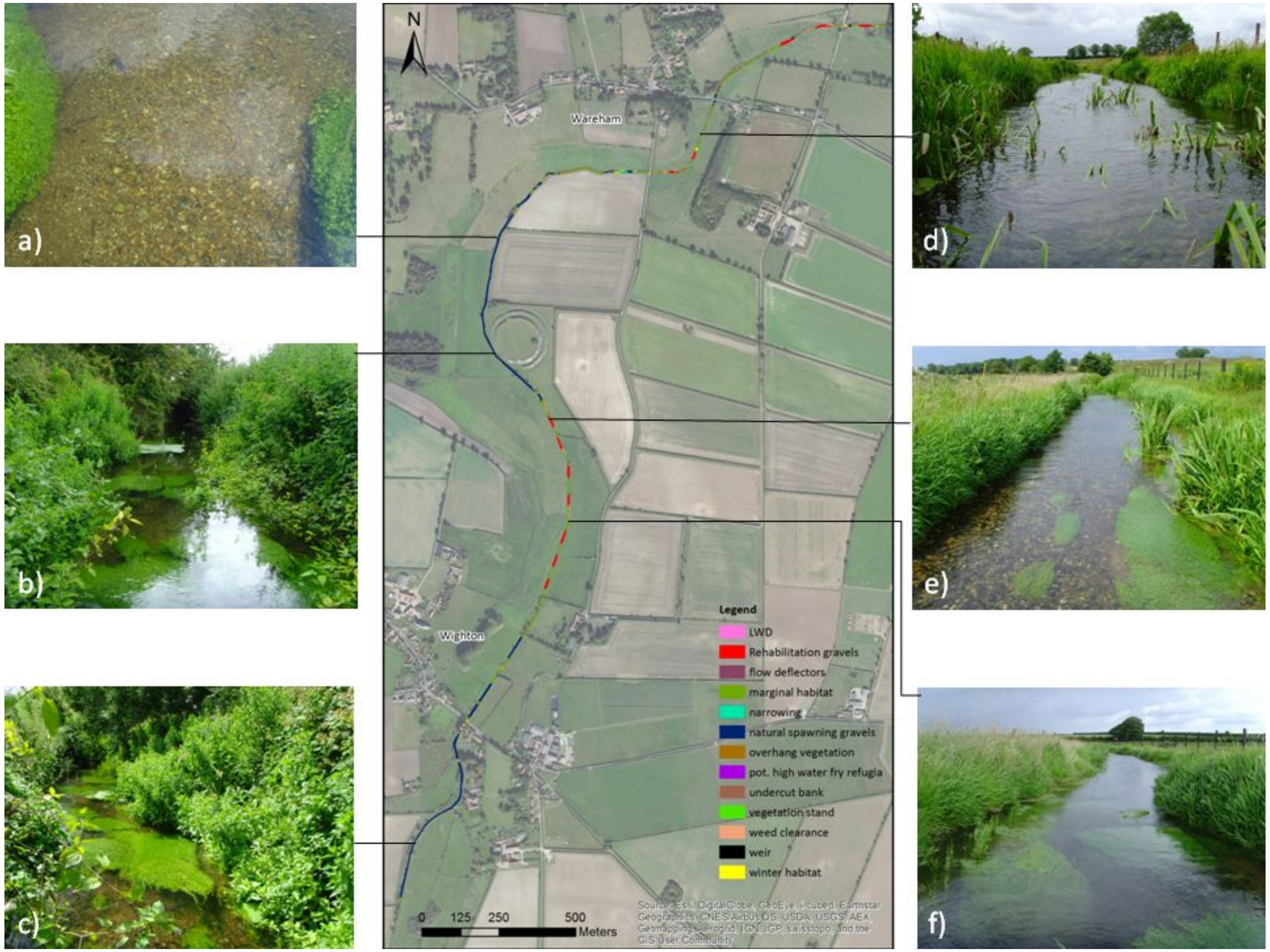



**Figure 6.5** Photos of wooded (a-c) and meadow (d-f) stream reaches distributed throughout the study site. Streambed gravels and shallower water depths were associated with wooded reaches, whilst rehabilitation gravel was associated with deeper water in meadow reaches. A typical streambed gravel (a), typical wooded reaches with good macrophyte growth (b) and c), the straight and deep nature of meadow reaches (d) and (f) as well as rehabilitation gravel within a meadow reach (e).

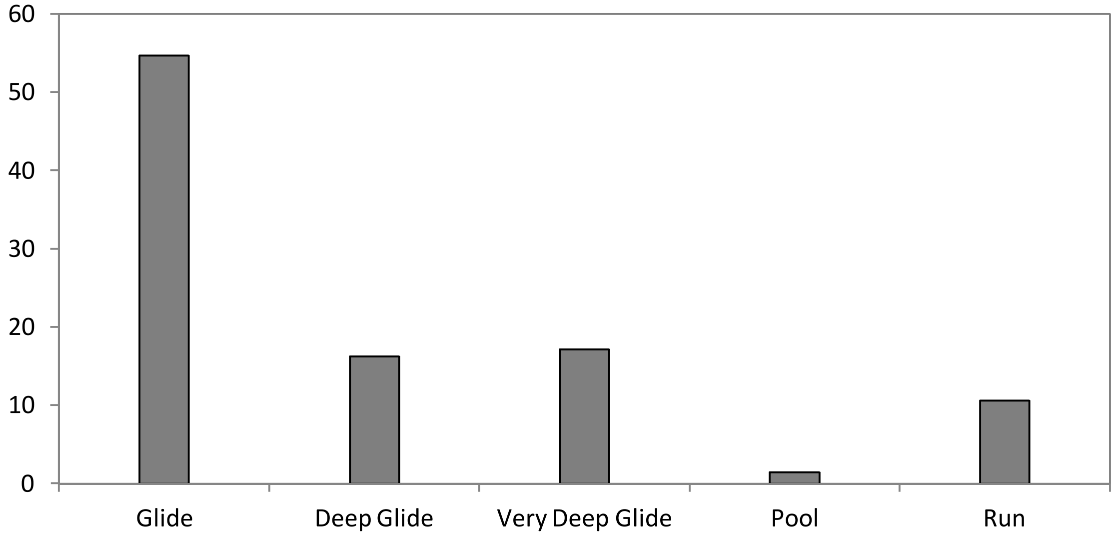



**Figure 6.6** Cumulative area ($m^2$) of different flow biotopes distributed throughout the study site. >80% of flow was one of the various glide biotopes. Pools and run biotopes contributed very little to the flow habitat.

**Table 6.4** Summary of key substrate characteristics per flow biotope. Channel processes were dominated by deposition and the composition of fine sediment (*D*<1 mm) on the streambed was high.

| **Flow Biotope** | **Predominant substrate characteristic** |
|---|---|
| Glide | Medium gravel, sand (16<*D*<0.05 mm) |
| Deep glide | Fine sediment (*D*<1 mm) |
| Very deep glide | Fine sediment (*D*<1 mm) |
| Pool | Fine sediment (*D*<1 mm) |
| Run | Mediam-coarse gravel (64<*D*<16 mm) |

**Table 6.5** Zonal analysis of percentage distribution of flow biotope over wooded and meadow reaches. Wooded reaches consisted only of glide and run flow biotopes. 100% of the deep glide, very deep glide and pool biotope were observed in the meadow reaches.

| Reach | Glides (%) | Deep glides (%) | Very deep glides (%) | Pools (%) | Runs (%) |
|---|---|---|---|---|---|
| Wooded-upstream | 22.2 | 0.0 | 0.0 | 0.0 | 33.5 |
| Wooded-downstream | 30.1 | 0.0 | 0.0 | 0.0 | 0.0 |
| Meadow-upstream | 27.8 | 38.2 | 20.9 | 60.9 | 37.5 |
| Meadow-downstream | 20.0 | 61.8 | 79.1 | 39.1 | 29.0 |
| **Wooded total** | 52.2 | 0.0 | 0.0 | 0.0 | 33.5 |
| **Meadow total** | 47.8 | 100.0 | 100.0 | 100.0 | 66.5 |

**Table 6.6** Zonal analysis of percentage distribution of flow biotope within each reach type. Wooded reaches comprised largely of the glide biotope whilst meadow reaches comprised of the various glide biotopes (deep, and very deep glide).

| Biotope | Wooded-upstream | Wooded-downstream | Meadow-upstream | Meadow-downstream | Wooded | Meadow |
|---|---|---|---|---|---|---|
| Glides | 77.2 | 100 | 50.8 | 28.6 | 88.9 | 38.3 |
| Deep glides | 0 | 0 | 20.8 | 26.4 | 0 | 23.9 |
| Very deep glides | 0 | 0 | 12.0 | 35.4 | 0 | 25.1 |
| Pools | 0 | 0 | 3.0 | 1.5 | 0 | 2.2 |
| Runs | 22.8 | 0 | 13.4 | 8.1 | 11.1 | 10.4 |
| **Total (%)** | 100 | 100 | 100 | 100 | 100 | 100 |

## 6.5 Distribution of early life-stage specific habitat

Abundance and area ($m^2$) of early life-stage specific habitat, accumulated every 250 m, indicated a mostly sparse longitudinal spatial distribution of habitat types throughout the study site (Figures 6.7-6.9). Marginal habitat and overhanging vegetation were, however, relatively abundant. Although fry refugia and rehabilitation gravel were typically associated with meadow reaches, the abundance of habitat types did not reflect wooded or meadow stream reach types, unlike flow biotopes. Wooded reaches were however characterised by shallow stream depths, good macrophyte growth as well as streambed gravels (Figure 6.5a-c), which occupied 73.3% of the total habitat area observed there (Table 6.7). Meadow stream reaches, on the other hand, were typified by straight deep channels, rehabilitation gravel and marginal habitat that contributed 36.8% and 27.3% respectively to the total habitat area ($m^2$)(Figure 6.5d-f, Table 6.7).

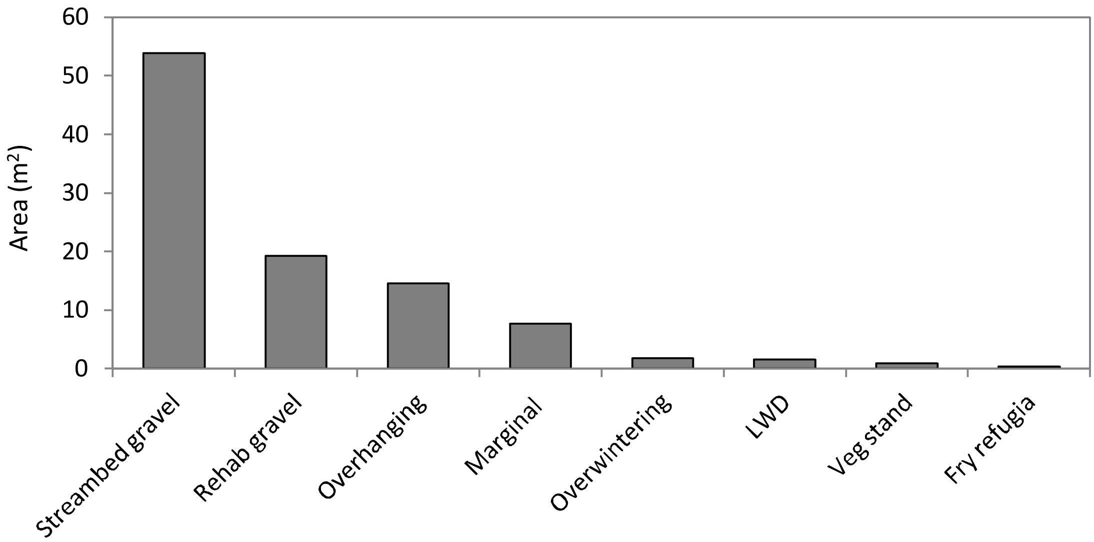



**Figure 6.7** Habitat types distributed throughout the study site based on area ($m^2$) coverage. Natural gravel occurred mainly in wooded reaches whilst all rehabilitation gravel was located in meadow reaches. There was a very low abundance of fry refugia, LWD and overwintering habitat.

The streambed gravel habitat type had the greatest contribution to the total cumulative early life-stage specific habitat area, 53.3%, of which 82.5% of the total area ($m^2$) occurred in wooded stream reaches (Figure 6.7, Table 6.8). Streambed gravels, however, included non early life-stage specific habitat and as such can provide a biased indication of natural spawning gravel abundance. Marginal habitat was ubiquitous throughout, but only contributed 7.7% to the total habitat area (Figure 6.7 and 6.8). Although marginal habitat abundance was higher in wooded reaches (Figure 6.8), >70% of the total marginal habitat area ($m^2$) was observed in meadow reaches (Figure 6.9, Table 6.8). 41.8% of the total marginal habitat area ($m^2$) was limited to the upstream meadow reach. This stream reach contained all 2003 rehabilitation gravels, as well as 2009A-D rehabilitation gravels.  Rehabilitation gravel constituted 19.2% of total habitat area ($m^2$) observed and were located in meadow reaches only (Figures 6.7-6.9). Rehabilitation gravel were installed into these over-deepened reaches (see section 3.3.2, Chapter 3).

Artificially steep river banks throughout the study site precluded an abundance of suitable shallow habitat areas for recently emerged fry. Steep banks and the straight nature of the channel also limited the frequency and extent of fry refugia suitable for occupation during high flow events. Fry refugia contributed only 0.3% of the total cumulative habitat area ($m^2$), all of which occurred in meadow reaches (Figures 6.8 and 6.9, Tables 6.7 and 6.8). 77.8% of the total fry refugia habitat occurred in the upstream-most meadow habitat (Table 6.8). Although overwintering habitat was sparsely abundant, 1.8% of total habitat area ($m^2$), most was observed within wooded reaches (Figures 6.7 and 6.8). However, a greater area coverage occurred in the meadow reaches, 69.7%, of which <67.4% occurred in the downstream most reach (Table 6.8). Large woody debris (LWD) contributed 1.6% to the total cumulative habitat area (Figure 6.7). Although LWD was observed in low abundance, wooded reaches contained a greater area contribution, 79.5%, indicating a high area per habitat polygon. Most of this total habitat area, 54.2%, was observed in a single upstream wooded stream reach (Table 6.8). The total area coverage of overhanging vegetation was slightly greater in wooded reaches than meadow reaches, 56% and 44% respectively (Table 6.8). Overhanging vegetation had a good contribution to the total overall habitat area ($m^2$), 14.6% (Figure 6.7). Further, marginal habitat removal took place throughout the meadow reaches only. This was a likely reflection of the easier accessibility to the river channel in these areas. Moreover, quantities of this habitat were removed from these reaches during the drought flow conditions of early 2012 and was a probable strategy to increase water conveyance for surface abstraction pipes further downstream.

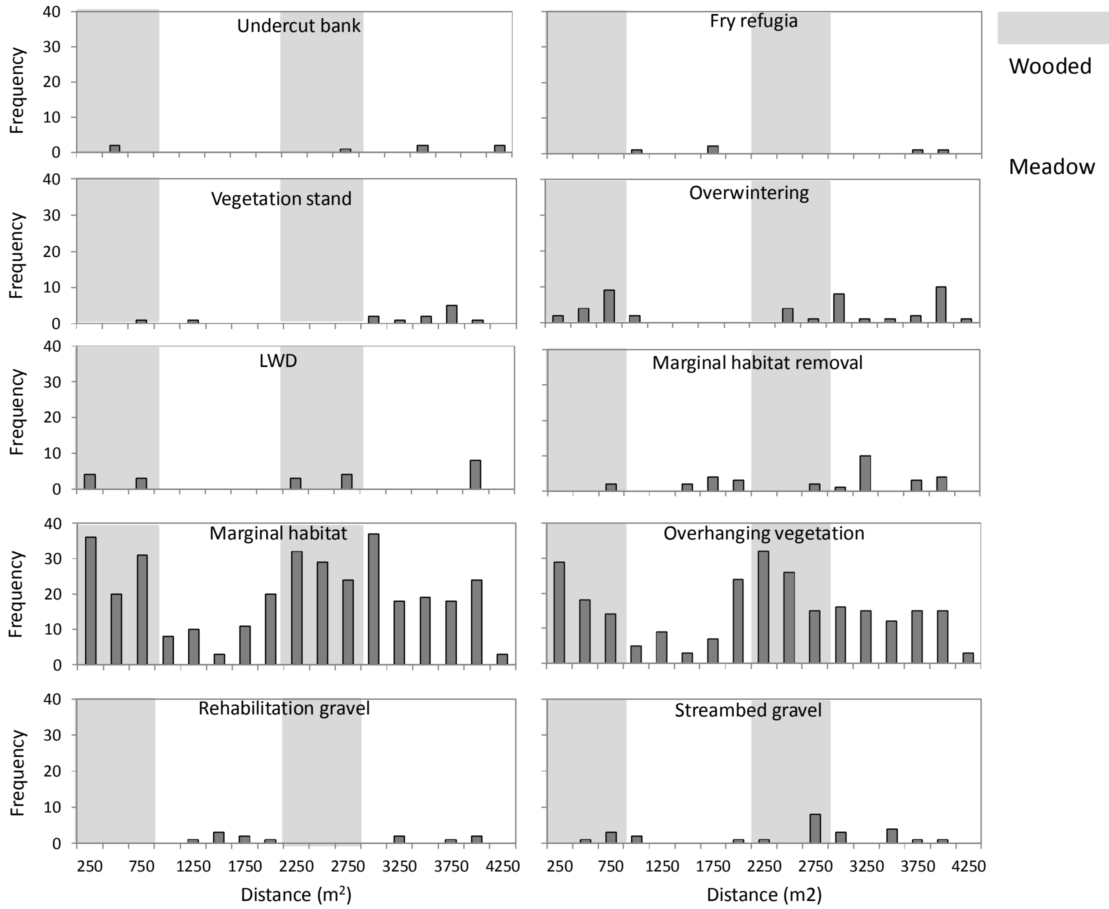



**Figure 6.8** Frequency histograms of early life-stage specific habitat types distributed throughout the study site. The centre point of each habitat polygon was plotted in 250 m bins measured from the upstream most point of the study site. Increasing distance is in a downstream direction from this point. Habitat types were largely characterised by low abundance. Many smaller habitat patches provide greater spatial coverage and thereby greater potential spatial connection between habitat types. However, larger habitat will often have a greater carrying capacity, depending on size of fish and habitat type. See Figure 6.9 for the distribution of cumulative habitat area ($m^2$) of each habitat type.

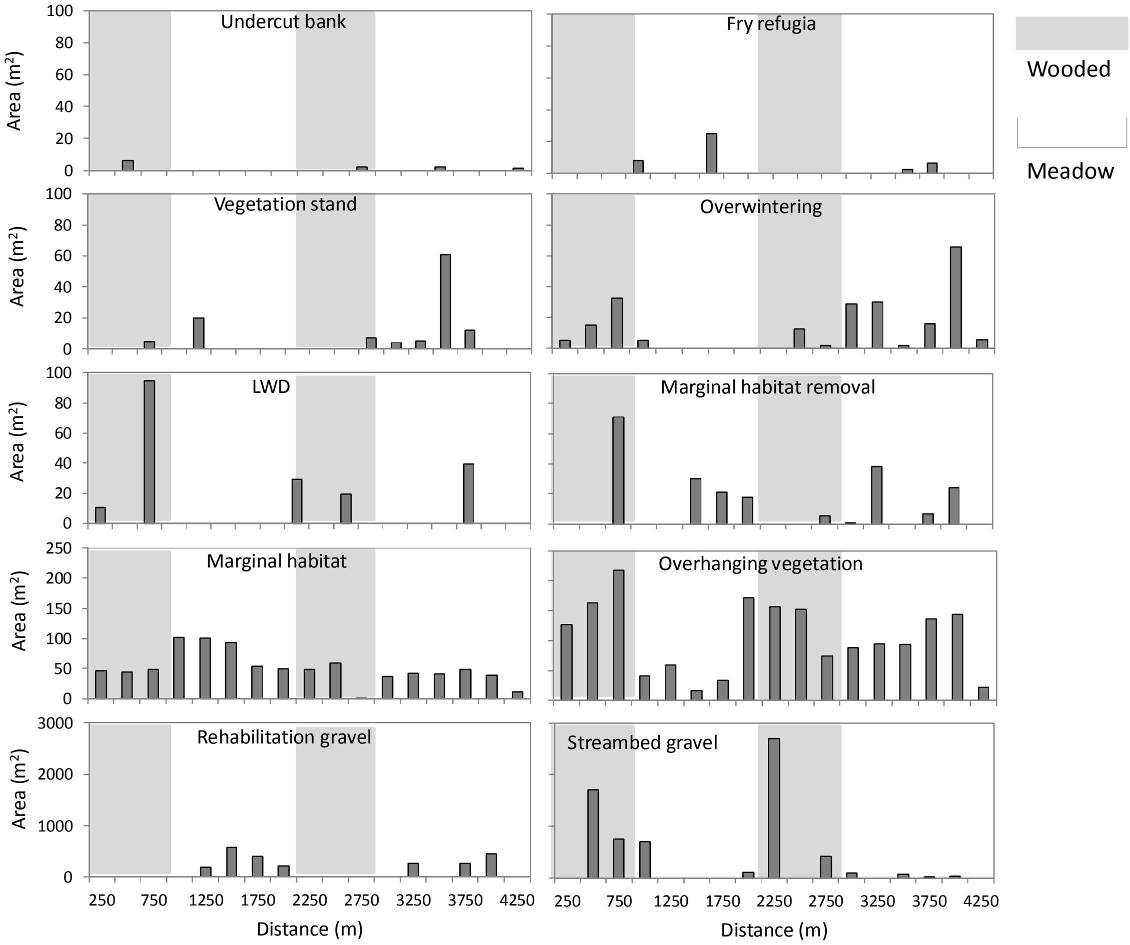



**Figure 6.9** Distribution of cumulative habitat area ($m^2$) of each early life-stage specific habitat type. The centre point of each habitat polygon was plotted in 250 m bins measured from the upstream most point of the study site. Increasing distance is in a downstream direction from this point. Habitat types every 250 m were largely characterised by low cumulative area ($m^2$). See Figure 6.8 for frequency distribution of each habitat type every 250 m.

**Table 6.7** Zonal analysis of percentage area ($m^2$) contribution of each life-stage specific habitat for wooded and meadow reaches. Wooded reaches were characterised by streambed habitat. Rehabilitation gravel was installed into meadow reaches only.

| **Reach** | **Wooded-upstream** | **Wooded-downstream** | **Meadow-upstream** | **Meadow-downstream** | **Wooded** | **Meadow** |
|---|---|---|---|---|---|---|
| Refugia | 0 | 0 | 1.0 | 0.3 | 0 | 0.7 |
| LWD | 3.0 | 1.2 | 0 | 1.3 | 2.1 | 0.6 |
| Marginal | 10.2 | 9.9 | 30.1 | 24.0 | 10.1 | 27.3 |
| Streambed | 70.1 | 76.1 | 23.5 | 11.6 | 73.3 | 18.0 |
| Overhanging | 14.8 | 12.1 | 4.8 | 21.1 | 13.4 | 12.3 |
| Overwintering | 1.5 | 0.3 | 0.1 | 5.0 | 0.9 | 2.4 |
| Rehab | 0 | 0 | 39.9 | 33.1 | 0 | 36.8 |
| Undercut Bank | 0.2 | 0.3 | 0 | 0.4 | 0.3 | 0.2 |
| Veg Stand | 0.1 | 0 | 0.6 | 3.1 | 0.1 | 1.7 |
| **Total (%)** | 100 | 100 | 100 | 100 | 100 | 100 |

**Table 6.8** Zonal analysis of percentage area ($m^2$) of early life-stage specific habitat types distributed over each wooded and meadow reach. Rehabilitation gravel and fry refugia were observed solely in meadow reaches. LWD had greater abundance in wooded reaches as did streambed gravels and overhanging vegetation. Meadow reaches had greater percentage of marginal, overwintering, and vegetation stand habitat types. Vegetation stand habitat comprised islands of emergent vegetation, typically *Sparganium erectum* (bur-weed).

| **Reach** | **Flow refugia (%)** | **LWD (%)** | **Marginal (%)** | **Streambed (%)** | **Overhanging (%)** | **Overwintering (%)** | **Rehab (%)** | **Undercut Bank (%)** | **Veg Stand (%)** |
|---|---|---|---|---|---|---|---|---|---|
| Wooded upstream | 0 | 54.2 | 14.3 | 37.2 | 29.1 | 24.0 | 0 | 22.8 | 4.5 |
| Wooded downstream | 0 | 25.3 | 15.6 | 45.3 | 26.7 | 6.3 | 0 | 38.2 | 0 |
| Meadow upstream | 77.8 | 0 | 41.8 | 12.3 | 9.4 | 2.3 | 58.7 | 0 | 17.5 |
| Meadow downstream | 22.2 | 20.5 | 28.3 | 5.2 | 34.8 | 67.4 | 41.3 | 39.0 | 78.0 |
| **Wooded total** | 0 | 79 | 30 | 82 | 56 | 30 | 0 | 61 | 5 |
| **Meadow total** | 100.0 | 20.5 | 70.1 | 17.5 | 44.2 | 69.7 | 100.0 | 39.0 | 95.5 |

## 6.6 Spatial relationship between juvenile *S. trutta* habitat: inference for population recruitment

Functional habitat units (FHU) defined areas where juvenile *S. trutta* production was high based on the spatial proximity between life-stage specific habitat. The maximum migration ability of juvenile *S. trutta* to move between habitat types constrained these areas. Life-stage specific habitat types for recently emerged fry were spawning gravel and marginal (nursery) habitat. The spatial relationship between these habitat types was examined at the maximum migration distances for low and high water levels, 10 m and 40 m respectively. Lengths of river channel where these habitat types occurred together within the constrained migration distances represented an area ($m^2$), defined by the specific habitat area ($m^2$), of population recruitment potential.

The spatial relationship between spawning gravel and marginal habitat reflected spatial fragmentation between wooded and meadow reaches in the River Stiffkey (Figure 6.10a and b). Juvenile *S. trutta* production area ($m^2$) was greater in the wooded reaches before and after rehabilitation gravel was installed, even though all rehabilitation was carried out in meadow reaches. Natural spawning gravel, surveyed within the streambed gravel habitat type, was spatially variable but abundant and therefore well spatially connected to marginal habitat. As such FHU were not spatially fragmented where streambed gravel occurred within the River Stiffkey. Fry recently emerged from these gravels had a greater chance of accessing marginal habitat and thereby contributing to population recruitment. However, the amalgamation of naturally occurring spawning gravels into the streambed gravel habitat type artificially exaggerated FHU in wooded reaches. Although >70% of the marginal habitat area ($m^2$) occurred in meadow reaches (Table 6.8), this was poorly spatially connected to rehabilitation gravel. Spatial fragmentation between habitat types in meadow reaches was however reduced after rehabilitation gravel was installed (Figure 6.10a-ii). Although rehabilitation gravel had a suitable spatial relationship with marginal habitat within the maximum migration distance, spatial fragmentation between rehabilitation gravel themselves reduced recruitment potential at the juvenile life-stage. The discrete and isolated nature of rehabilitation gravel created a series of small and fragmented *S. trutta* production zones in meadow reaches, observed as a series of small area ($m^2$) FHU peaks (Figure 6.10a-ii and b-ii). There was greater utilisation of marginal habitat within wooded stream reaches for *S. trutta* production that was not replicated in meadow reaches.

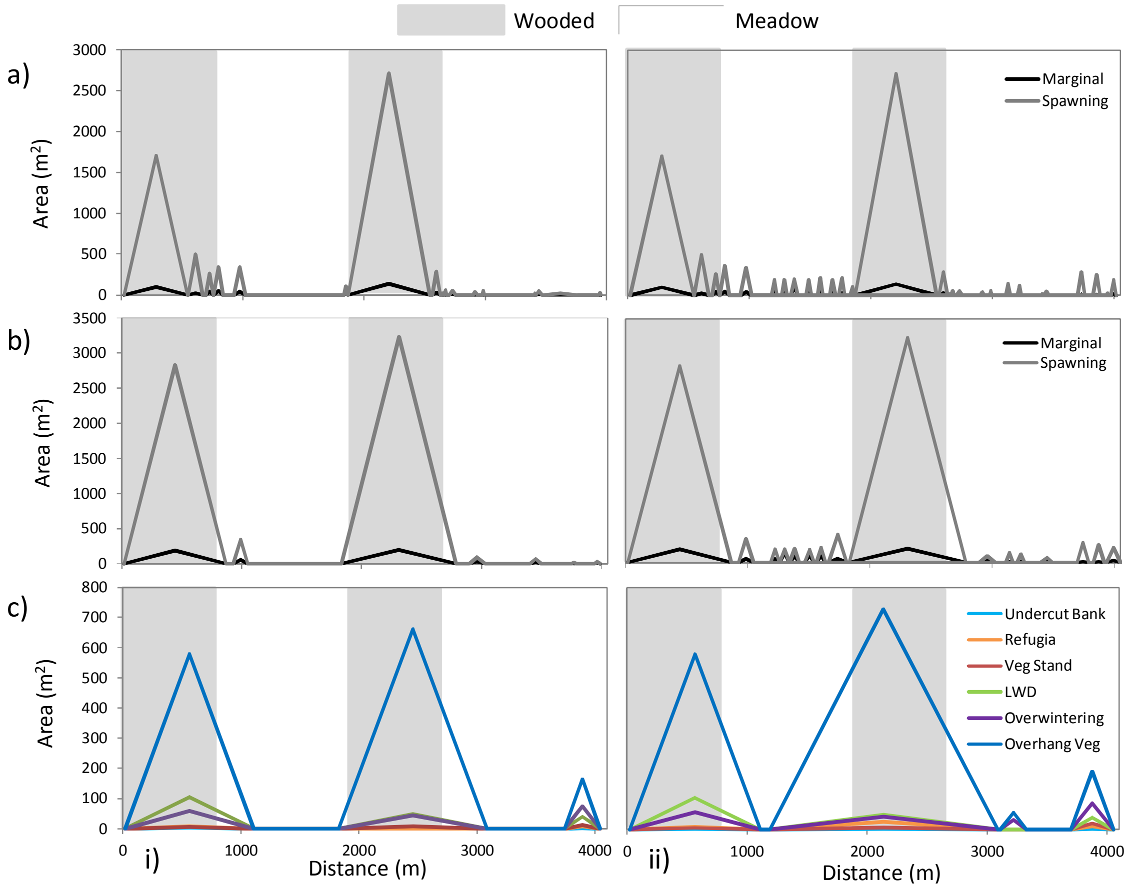



**Figure 6.10** Functional Habitat Units (FHU) plots for the juvenile stage of *S. trutta*, a) at 10 m, b) at 40 m, and c) at 100 m, prior to rehabilitation (i) and post rehabilitation (ii). Wooded and meadow reaches are denoted by shading or no-shading respectively within each plot. Stream distance is measured along the *x*-asis and as such indicated the length distribution of each FHU. The location along the channel length and area ($m^2$) of those habitat types that occur within suitable migration distances (10-40m, 100 m) of each other to contribute towards juvenile *S. trutta* produciton are plotted as FHU. FHU are spatially fragmented throughout the length of the study site, and reflected wooded and meadow stream reaches that were defined by historic dredging activity. Rehabilitation gravel were observed as distinct spatially fragmented peaks in meadow reaches. FHU were only increased during winter months by the addition of rehabilitation gravel (c).

At greater stream flow a migration distance of 40 m from spawning gravels to marginal habitat increased the FHU through wooded reaches more than within meadow reaches (Figure 6.10a-ii and b-ii). While an increased migration distance (40 m) reduced spatial fragmentation within meadow reaches, and connected two rehabilitation gravel sites (2009C and D) in the upstream reach into a single larger FHU area ($m^2$) with an associated greater juvenile production potential, the reduced spatial fragmentation between streambed gravel areas were more readily assimilated under the greater migration distance. Reduced spatial fragmentation between individual rehabilitation gravel sites would have resulted in larger but fewer FHU areas in the presence of suitable and accessible marginal habitat.

Life-stage specific habitat types for 0+ parr (older fry within first year) included undercut banks, large woody debris (LWD), vegetation stands and overhanging vegetation. As 0+parr remain close to their natal spawning gravels (Solomon and Templeton, 1976; Bachman, 1984; Armstrong et al., 1994), overwintering FHU were determined using a maximum migration distance of 100 m downstream of spawning gravels. Overwintering FHU included the area ($m^2$) of life-stage specific habitat types for 0+ parr. However, if the 100 m migration distance did not contain overwintering habitat, then that FHU was not considered. Overwintering FHU within the study area also reflected the spatial fragmentation between wooded and meadow reaches in the River Stiffkey (Figure 6.10c-i and ii). Prior to rehabilitation (Figure 6.10c-ii), overwintering FHU were spatially fragmented into three distinct zones each with good habitat diversity. The larger two zones reflected wooded stream reaches through primarily the presence of good overhanging vegetation cover. Gravel rehabilitation reduced overwintering FHU fragmentation and increased the spatial proximity to 0+ parr life-stage specific habitat through most of the length of stream. Overwintering habitat was however relatively sparse throughout most of the study reach (Figures 6.8 and 6.9) reaching approximately 100 $m^2$ at its greatest extent in the downstream meadow stream reach (Figure 6.10c-i and ii). Although the downstream-most meadow reach contained the greatest area ($m^2$) of overwintering habitat, <70%, spatial fragmentation between spawning gravels and overwintering habitat reduced the *S. trutta* recruitment potential in this reach.

## 6.7 Discussion

### 6.7.1 Habitat bottlenecks to population recruitment at the juvenile life-stage

Habitat is a key driver of population recruitment in chalk streams where there are typically good food resources (Mann et al., 1989) and low temperature and flow variability. *S. trutta* populations exhibit a strong size structured spatial distribution associated with preferential habitat use (Baran et al., 1997; Heggenes et al., 1999). There is a direct relationship between population density and sub-optimum habitat use (Greenberg, 1994). Embedded within this relationship, larger fish outcompete smaller fish for optimum habitat space. Spatial fragmentation is therefore driven by intraspecific competition, and consequentially determined by population density (Baran et al., 1997). Juvenile *S. trutta* (alevin and parr) are aggressively territorial; they form hierarchies, compete for resources and defend territories (Heggenes et al., 1999; Klemetsen et al., 2003). Increases in juvenile abundance therefore increase competition for suitable habitat and food resources (Bohlin, 1977; Greenberg 1994; Milner et al., 2003). Elliott (1994) observed that those juveniles which fail to successfully secure suitable habitat have a greater mortality risk. Availability of habitat therefore regulates *S. trutta* population recruitment through these density-dependent factors at the early life-stages (Elliott, 1989; Armstrong et al., 2003; Milner et al., 2003).

Furthermore, population recruitment is inversely related to the temporal and spatial variability of life-stage specific habitat (White, 1999; Kocik and Ferreri, 1997). Greater salmonid production occurs in stream reaches where there is little spatial fragmentation between habitat types, and lower in those reaches with high spatial fragmentation. *S. trutta* are sedentary throughout much of the year preferring not to move long distances (Knouft and Spotila, 2002). The spatial proximity of key habitat types suitable for different life-stages, particularly juvenile habitat such as between spawning gravels and marginal habitat, are therefore an important determinant of population health (Huusko et al., 2007). If the spatial proximity between habitat types exceeds feasible migration distances of 10-40 m for recently emerged alevins, or 100 m for overwintering habitat, then mortality at that life-stage increases. Spatial habitat fragmentation therefore represents a physical bottleneck to population recruitment. There is therefore a requirement on the management process to be sympathetic of the spatial scale and distribution of other key life-stage specific habitat types when installing rehabilitation gravel. Introduction of rehabilitation gravel in this respect increased abundance of spawning gravel but did not effectively address rehabilitation objectives to improve population recruitment.

### 6.7.2 Spawning and nursery habitat in the River Stiffkey

*S. trutta* are typically limited by the availability and suitability of spawning and nursery habitat in chalk streams (Mann et al., 1989). In the River Stiffkey, juvenile *S. trutta* production is regulated by the poor spatial connectivity between spawning gravel and marginal nursery habitat, particularly at low stream flow (Figure 6.10 a and b). Wooded stream reaches have greater juvenile production potential as spatial connectivity and habitat diversity are greatest (Figure 6.10). Poor habitat spatial connectivity is more prevalent in meadow reaches where anthropogenic activities, such as dredging, have had a greater effect on channel characteristics

(Figure 6.5d and f). Fry emerging from rehabilitation gravel have a greater likelihood of not securing suitable habitat and therefore not contributing to population recruitment in these reaches (Figure 6.10a-ii and b-ii). There is strong evidence in the literature for a density-dependent period of increased mortality after emergence termed the 'critical period' after which mortality rates decrease (Elliot, 1988; Egglishaw and Shackley, 1977). Juvenile *S. trutta* have greater susceptibility to predation than their larger conspecifics. There is therefore a survival advantage in establishing optimal habitat with access to an abundant food resource as early as possible for increased growth rates (Elliott, 1989). Young fry not able to secure suitable habitat face greater mortality risk (Elliott, 1989). Augmentation of the *S. trutta* population by means of introducing rehabilitation gravel only (with no other measure) has a reduced potential for increasing population recruitment based on the spatial relationships between juvenile specific habitat. Installation of rehabilitation gravel designed to augment *S trutta* populations should address marginal habitat abundance within close proximity to rehabilitation gravel in order to reduce density-dependent intraspecific competition and consequentially emergent fry displacement (Daufresne et al., 2005). It is anticipated that rehabilitation gravel installed with reduced spatial fragmentation in a less discrete and isolated manner with a greater association with marginal habitat would yield greater juvenile *S. trutta* production.

### 6.7.3 Overwintering habitat: effects of poor abundance on population recruitment

Overwintering is an environmentally defined life-stage (Hurst, 2007) critical to population recruitment (Elliott, 1989). There are multiple physical and biological factors that regulate winter mortality of juvenile fish: abundance and suitability of refugia, severity of and exposure to thermal stress, fish energy reserves and starvation, as well as exposure to predation risk (Hurst, 2007; Huusko et al., 2007). *S. trutta* exhibit a narrower spatial habitat niche in winter than they do in summer (Lund et al., 2003; Heggenes et al., 1993; Heggenes et al., 1999). Optimal winter habitat reduces the need for energetic expense, thus reducing potential energetic deficits (Huusko et al., 2007). Suboptimal habitat increases the requirement of *S. trutta* to catabolise energy reserves (fats, proteins and glycogen) (Cunjak and Power, 1987). The availability of suitable overwintering refuge space is key to the survival of juvenile salmonids (Heggenes et al., 1993; Harwood et al., 2002; Hurst, 2007). Winter months bring conspecifics into direct competition for available habitat. Refuge sharing during winter is not common (Cunjak and Power, 1986; Mitro and Zale, 2002) and limited habitat availability is associated with increased intra- and inter-specific competition and (mammalian and ovarian) predation risk (Armstrong and Griffiths, 2001; Harwood et al., 2002; Hurst, 2007). Larger fish outcompete smaller conspecifics for habitat space (Harwood et al., 2001). As such habitat availability and the narrow suitability selection criteria can increase intraspecific competition during winter months. Winter mortality and population recruitment is linked to habitat abundance and associated inter- and intra-specific competition (Annear et al., 2002). Lund et al. (2003) observed an increase in juvenile (0+) *S. trutta* mortality during winter in response to increased intraspecific competition. In their study on two rivers in northern Finland, the Rivers Kuusinkijoki and Kitkajoki, Mäki-Petäys et al. (1999) determined that a lack of suitable habitat during winter can create a juvenile (0+) *S. trutta* bottleneck to recruitment.

Available habitat within suitable migration distances reduces predation risk and the requirement to burn energy (Heggenes et al., 1993; Armstrong and Griffiths, 2001). Overwintering habitat refuge in the River Stiffkey was not spatially fragmented exhibiting a suitable spatial proximity (<100 m) from summer habitat throughout the study site (Figure 6.10c). Migration distances were therefore low and consequently a reduction of intraspecific competition and mortality would follow (Huusko et al., 2007). However, overwintering habitat had a low abundance in the River Stiffkey. Similar comparisons with other rivers, including chalk streams, are limited. Although many juvenile *S. trutta* winter mortality studies advocate the addition of suitable overwintering habitat (Muhlfeld et al. 2001; Dare et al., 2002), quantitative investigation of overwintering habitat remain scarce. Based on studies conducted by Cunjak and Power (1986), Greenberg (1994), Armstrong and Griffiths (2001) and Harwood et al. (2002) into intraspecific sharing of overwintering habitat, it is anticipated, based on the aggressive competition for available habitat, that juvenile *S. trutta* in the River Stiffkey will have increased vulnerability during winter. Due to the low abundance of overwintering habitat and the associated intraspecific competition for habitat resources, it is likely that a size-structured spatial segregation exists amongst the *S. trutta* population. Furthermore, it is likely that the winter months are a potential bottleneck to recruitment in the River Stiffkey (Armstrong and Griffiths, 2001; Hurst, 2007).

### 6.7.4 Spatial scale of stream management

Stream management frequently disregards spatial scales of rehabilitation even though such management is key to the success of *S. trutta* carrying capacity (Heggenes et al., 1999). Functional habitat units (FHU) are an indispensible tool used to investigate the natural spatial scale at which management is required to improve population recruitment (Kocik and Ferreri, 1997). They provide an indication of the availability and spatial scale of population recruitment zones along a selected river channel segment. The spatial relationship between key life-stage dependent habitat defines the natural scale at which to manage population production. FHU defined at 10 m, 40 m and 100 m indicated spatial fragmentation at each of these distances that divided the study site into 4 distinct reaches.

Over-deepened meadow reaches had poor juvenile *S. trutta* production potential. Although installation of rehabilitation gravel in these reaches was aimed at addressing poor recruitment, the natural spatial scale between early life-stage specific habitat types appeared to have been overlooked. Hurst (2007) argued that despite strong scientific evidence for high rates of juvenile winter mortality, management strategies rarely increased overwintering habitat suitable for all life-stages of *S. trutta*. In the River Stiffkey overwintering habitat was observed in low abundance (Figure 6.8 and 6.9). Rehabilitation gravel reduced, to some degree, habitat fragmentation but inadequately addressed connectivity of juvenile life-stage dependent habitat at the appropriate reach scale. This further fragmented habitat availability throughout meadow reaches into isolated zones of poorly connected juvenile habitat. Management strategies focused on increasing the relative abundance of necessary juvenile habitat at the correct spatial scale whilst opening up new spawning habitat will reduce population stress at the juvenile life-stage. Management effort should therefore provide suitable and accessible juvenile habitat in addition to rehabilitation gravel at a minimum of a 10 m scale for recently emerged *S. trutta* fry and overwintering habitat at a 100 m scale to reduce intraspecific competition.

### 6.7.5 An alternative habitat rehabilitation strategy

Activities such as channelisation and flood mitigation management have all contributed to a reduction in the natural range of *S. trutta* (Acornley and Sear, 1999; Hendry et al., 2003; Pedersen et al., 2009). Dredging of the streambed is the principal cause of morphological degradation (Brookes, 1986; Petersen et al., 1992; Gregory and Davis, 1997). River channel dredging has had discernible impacts on *S. trutta* recruitment in the River Stiffkey dividing the study site into two reach types of contrasting juvenile production potential (Figure 6.10). The river channel has been appreciably deepened through the two meadow stream reaches for flood mitigation management. This has considerably altered the physical character of the streambed, removing streambed gravel and homogenising the sediment composition towards finer grained sediment sizes. Sediment derived from run-off (see section 3.2.2, Chapter 3) is stored largely within these meadow reaches in response to localised reductions in stream velocity as a consequence of channel modification. River channel deepening has simplified the hydraulic regime in meadow reaches with subsequent implications on hydrogeomorphic processes that have altered sediment transport capacities and encouraged depositional processes to dominate (Brookes, 1985; Fryirs and Brierley, 2012; Landemaine et al., 2015). Furthermore, the low stream power, characteristic of chalk streams, makes the loss of gravel streambed habitat irreplaceable (Mainstone et al., 1999).

Salmonid populations are dependent on the availability and suitability of habitat (Heggenes et al., 1999; Armstrong et al., 2003). The global decline of salmonids is in part due to a loss of suitable habitat as anthropogenic demands on water resources increase. As population carrying capacity and density are regulated by habitat diversity and suitability (Bohlin, 1977; Milner et al., 1985; Heggenes et al., 1999; Klemetsen et al., 2003), habitat rehabilitation has become a key management strategy for improving salmonid stocks (Hendry et al., 2003). Assessment of life-stage specific habitat is therefore an important management requirement (de Jalón, 1995; Maddock, 1999). Habitat surveys are an essential management tool for investigating limitations imposed on populations, as they provide a quantitative assessment of the potential carrying capacity of a stream (Milner et al., 1985). Because juvenile populations are readily constrained through biotic and abiotic factors, investigation of the early life-stages are of critical importance for population dynamics and salmonid management. There are few, if any, juvenile *S. trutta* habitat survey results from chalk streams available in the wider scientific literature. Such data likely remains in the domain of grey literature and as such not widely available.

Spatial habitat fragmentation limits juvenile production potential of meadow reaches in the River Stiffkey, even after rehabilitation gravel had been installed. Indeed, given the longevity of rehabilitation gravel for embryo development (see section 5.9.7, Chapter 5), the over-deepened dredged reaches will remain of very limited value to juvenile *S. trutta* production. Given the deep character and sediment deposition dominated state of the meadow reaches, installation of rehabilitation gravel within wooded reaches would have been a preferable alternative. Greenberg (1994) noted that shallow water over medium gravel ($16<D<8$ mm) with faster stream flow, such as riffles and runs, were the preferred flow refugia for juvenile *S. trutta*. The River Stiffkey had few fast flow biotopes with the majority of faster water over shallow streambed gravels observed in wooded reaches (Figure 6.6, Tables 6.5 and 6.6). Wooded reaches also had greater macrophyte cover (Figure 6.5b and c). McRae (2005) noted that percentage gravel substrate and macrophyte abundance were key variables correlated with juvenile *S. trutta* density in the Au Sable River, Michigan, USA. Furthermore, overhanging vegetation, essential cover refugia for juvenile *S. trutta* (Heggenes et al., 1999), was observed in relatively high abundance over both wooded reaches but only the downstream most meadow reach (Figures 6.8 and 6.9, Tables 6.7 and 6.8). Egglishaw and Shackley (1982), Milner et al. (1985) as well as Heggenes (1996) observed a positive correlation between quantity of overhanging vegetation and juvenile *S. trutta* dispersal and concluded that this habitat type was a key control of abundance. This is likely due to the associated reduction in predation risk (Armstrong et al., 2003; O'Connor and Rahel, 2009). Hunt (1977) demonstrated a direct relationship between cover and population abundance; where cover had been introduced to a stream reach, abundance in that reach increased, whilst a reduction in abundance was related to loss of cover.

The over-deepened morphological condition of meadow reaches provide a more suitable cover refugia for adult *S. trutta*. Availability of deep water cover, specifically in small streams (Heggenes et al., 1999), is a key habitat variable for adult *S. trutta* salmonid abundance (Armstrong et al., 2003; Ayllón et al., 2010). Larger (more dominant) fish select deeper slower flowing habitats whilst juvenile fish occupy shallower habitats (Bohlin, 1977; Baran et al., 1997; Heggenes et al., 1999; Ayllón et al., 2010). Although rehabilitation gravel provide a refugia for juvenile fish, the deeper water adjacent to each rehabilitation gravel increase the predation risk to juvenile fish. The abundance of shallow streambed gravels and marginal habitat in wooded reaches form ideal juvenile nursery habitat and indicate that, whilst the potential for embryos to develop into fry might be great, the potential for fry to mature into adult fish is low due to the lack of deeper water in these reaches. Individual fish are required to migrate into meadow reaches to find deeper, more suitable habitat to facilitate further growth.

Reintroduction of natural stream processes, flow variability and streambed heterogeneity to the wooded reaches would increase population recruitment potential. Large woody debris (LWD) increases habitat abundance and complexity, offers refuge from predation and forms suitable overwintering habitat (Zika and Peter, 2002). LWD alters localised flow dynamics reintroducing natural stream processes, increasing streambed heterogeneity and scour of streambed gravels that form suitable *S. trutta* spawning habitat (Stewart et al., 2006). Like the River Stiffkey, much LWD has been cleared from river channels due to perceived increased risk to flood and erosion (Lester and Wright, 2009). Most (<80%) LWD was observed in wooded reaches (Figures 6.8 and 6.9, Tables 6.6 and 6.7) reflecting the difference in management strategies between these reach types; meadow reaches receive more channel maintenance than wooded reaches. LWD increases diversity in stream velocity without an increase to flood risk (Lester and Wright, 2009). Johnson et al. (2005) observed that LWD played a key role in increasing the survival and abundance of juvenile *S. trutta*. In-stream experiments conducted in the Mühlebach stream situated in Liechtenstein, Central Europe, have illustrated a positive correlation between the abundance of LWD and *S. trutta* abundance and biomass (Zika and Peter, 2002). Furthermore, in their review of 127 studies, Stewart et al. (2006) concluded that woody debris increased the abundance of *S. trutta* at the population level. Increasing the abundance of LWD in wooded reaches as a cheap and sustainable rehabilitation method would have considerable benefits for *S. trutta* population recruitment in the River Stiffkey.

### 6.7.6 Stream management: the importance of looking after our fringes

Improvements in the quality and abundance of marginal habitat increases food resources, nursery habitat, reduces inter- and intraspecific competition and predation risk, as well as regulating stream temperature and velocity (Zalewski and Gronkiewicz, 1998). Although in-stream rehabilitation structures increase *S. trutta* habitat abundance and complexity in channelised stream reaches (van Zyll de Yong et al., 1997), improvements to the marginal habitat did not occur in combination with gravel introduction on the River Stiffkey. Marginal habitat was actively removed from parts of the meadow reaches in February 2012, a period when this habitat would be most sought after by recently emerged fry seeking marginal habitat refugia. This was likely a response by farmers to increase water conveyance for a surface abstraction point further downstream. Loss of marginal habitat in rivers, through channelisation, excessive management and overgrazing, has negative implications on *S. trutta* production largely through a decline in juvenile fish abundance (Cunjak and Power, 1986; Summers et al., 2005; Riley et al., 2006). During studies on small Danish streams Mortensen (1977a) observed elevated fry mortality rates in areas of vegetation removal relative to undisturbed stream reaches. Moreover, Mitro and Zale (2002) concluded that overwintering survival increased by >20% in those stream reaches where complex bank habitat was abundant. Reduction of marginal habitat limits *S. trutta* recruitment in chalk streams (Mann et al., 1989).

### 6.8 Conclusions

Installation of rehabilitation gravel is a management method whereby *S. trutta* abundance can be enhanced through improved recruitment to the population. However, a suite of key life-stage specific habitat, other than spawning gravels, are required to achieve this objective. Should any of these habitat types occur in poor abundance, or be spatially fragmented within the river channel, population recruitment will remain low. As habitat requirements differ between life-stage and season, there is a need for surveys to focus on a particular life-stage in order to derive data that is useful at a management level (Milner et al., 1985). Identification through surveys of the deficiencies in key life-stage dependent habitat is essential to successful river management and rehabilitation strategies for salmonids (Hendry et al., 2003). The *S. trutta* population in the River Stiffkey is stressed at the juvenile stage: poor juvenile habitat connectivity and a low abundance of overwintering habitat increase population density stresses such as intraspecific competition. Juvenile *S. trutta* recruitment was therefore limited by low abundances and poor spatial relationships between key habitat types. There is a need to address rehabilitation schemes through an understanding of the importance of spatial connectivity of life-stage specific habitat at the reach scale. Juvenile production in the River Stiffkey can achieve greater potential under a focused management strategy that incorporates such measures. Rehabilitation gravel, however, could have been better situated in wooded reaches where juvenile production has greater potential than in meadow reaches. Additional habitat rehabilitation would have considerably improved potential population recruitment, specifically by increasing abundance of LWD in wooded reaches.

## 7 Discussion and Conclusions

The fundamental factors required for effective salmonid conservation are good water quality and a habitat complex that provides access to spawning gravel, a food resource and refuge from predation and elevated water stages (Armstrong et al., 2003; Hendry et al., 2003). Despite their protected status, UK chalk streams have suffered considerably from water quality issues and habitat degradation over recent decades. Excessive sediment input, in addition to channelisation and dredging, are responsible for significant habitat loss, which has had negative consequences for *S. trutta* production (Theurer et al., 1998; Greig et al., 2005a; Zimmermann and Lapointe, 2005; Hartman and Hakala, 2006). Conservation management efforts concerned with re-introducing *S. trutta* habitat, specifically those habitats for spawning and juvenile stages, are vital to population recruitment due to the very specific criteria for each vulnerable life-cycle stage. Juvenile habitat requirements differ from those of mature fish, which, due to some degree of habitat flexibility, are relatively non-specific. Moreover, successful recruitment to the *S. trutta* population is reliant on several life-stage dependent habitat types that persist within a suitable spatial proximity of one another. Spatial proximity between habitat types is determined by life-stage dependent migration capabilities. A management strategy that increases the abundance of spawning gravel without increasing other key life-stage dependent habitat effectively transfers existing bottlenecks from spawning habitat to habitat required for later life stages such as nursery and rearing life-stages.

The field of river rehabilitation is increasingly gaining recognition as an applied science suitable for addressing habitat decline in rivers, and the introduction of gravels to supplement salmonid spawning habitat is an increasingly adopted rehabilitation management strategy (Harper et al., 1998; WTT, 2008; Pulg et al., 2013). Rehabilitation gravel intended to improve migratory *S. trutta* population recruitment were introduced to the River Stiffkey in 2003 and again in 2009 by the Wild Trout Trust (WTT); the latter introduction was part of the Living North Sea (LNS) Project. Two mechanisms of achieving this objective were outlined; 1) encourage spawning habitat accessibility through increased passage between estuary and river, and 2) improve spawning habitat for *S. trutta* in the river. Studies investigating the quality of rehabilitation gravel are limited (see Barlaup et al., 2008; Pedersen et al., 2009), however, and very few studies have gone beyond quantifying short-term successes to examine the complex nature of localised physical controls on recruitment at the embryo life-stage or the key role in juvenile survival. There is some evidence to suggest that gravel introduction to rivers can be an effective in-stream rehabilitation tool for *S. trutta* recruitment. For example, Merz and Setka (2004) observed spawning on rehabilitation gravel 2 months after introduction on the Mokelumne River, California. Over a 30 month monitoring period the rehabilitation gravel increased water velocities, intragravel permeability and dissolved oxygen. However, Pulg et al. (2013) reported that *S. trutta* embryo survival in rehabilitation gravel installed in the Moosach River, southern Germany declined significantly over 6 years. The authors concluded that catchment-derived fine sediment input deposited in rehabilitation gravel caused a morphosedimentary change and the suitability of rehabilitation gravel for spawning degraded rapidly as a result (Pulg et al., 2013).

This study on the River Stiffkey suggests that, dependent on factors such as rainfall, sediment-in-wash and flooding, rehabilitation gravel will degrade into a completely unsuitable state for *S. trutta* spawning and embryo development in <10 years. This is because the River Stiffkey is sediment supply controlled; high inputs of fine sediment derived from soil erosion during high rainfall events and a poor sediment transport competence characterise the river. Historic channel modification such as dredging and straightening have impacted stream velocity variability creating flow homogeneity and enhanced sediment deposition. Installation of rehabilitation gravel in over-deepened and straightened reaches likely increased susceptibility to sediment deposition. Due to the readily available sediment supply and loss of stream velocity heterogeneity, rehabilitation gravel in the River Stiffkey undergo a succession from a very well sorted gravel type, with a narrow range of coarse gravel, towards a deposit characterised by a bimodal grain-size distribution, as excessive fine sediment alters the composition. The large interstitial voids, characteristic of Stiffkey rehabilitation gravel, accumulate an abundance of fine grained sediment from the bottom up. This reduces interstitial permeability and subsequently interstitial water velocities and DO delivery that inhibit embryo development.

*S. trutta* population recruitment in the River Stiffkey is controlled largely by spatial and temporal scales of catchment-derived sediment input, resulting in a vulnerable population reliant on a small number of spawning gravel sites. Rehabilitation gravel undergoes a physical morphosedimentary succession determined primarily by temporal scales of sediment input, and secondly through the spatial proximity to sources of sediment input. Fine grained sediment deposition altered the composition of spawning substrata resulting in a decline of embryo survival over the short-term. Poor land-use management and variation in precipitation had a considerable impact on this relationship. River rehabilitation based on the sound principles of geomorphology, not just at the reach scale but at the larger catchment scale, will have increased ecological benefits and greater sustainability.

The aims of LNS in the River Stiffkey were to improve the freshwater phase of the migratory *S. trutta* life cycle through enhanced population recruitment. However, installation of rehabilitation gravel in isolation from other key life-stage habitat was a further indication that this objective has not been successfully addressed. Functional habitat improvement for early life-stages should not focus on any single habitat type, but rather a suite of interconnected habitat types to accommodate the requirements at each stage during the first 6-8 months. The *S. trutta* population in the River Stiffkey is likely stressed at the juvenile stage due to poor connectivity between spawning gravel and marginal nursery habitat. Further, a low abundance of overwintering habitat increases intraspecific competition (Armstrong and Griffiths, 2001). In addition to poor embryo survival, the abundance of and spatial fragmentation between key life-stage dependent habitat types is a key variable that prevents rehabilitation gravel from providing a sustainable solution to population recruitment in the River Stiffkey.

## 7.1 Lessons learnt: applications for management

The River Stiffkey is susceptible to catchment-derived sediment inputs, typical of many chalk stream catchments, that limits the spawning value of rehabilitation gravel. Hydrogeomorphic processes have become sediment-supply controlled; accumulation of fine grained sediment is a function of supply exceeding the prevailing hydraulic conditions to erode and transport sediment loadings. Sediment-laden run-off from arable fields and livestock poaching of river banks remain concerns throughout the catchment. Bradshaw (1996) argued that in-stream management efforts should only be delivered once the underlying causes of environmental constraint have been identified and addressed, particularly those system controls that have little or no capacity for natural recovery. Management efforts on the River Stiffkey, for example, should therefore focus on altering hydrogeomorphic controls from a sediment-supply dominated state to one with greater emphasis on system stability. This can be achieved by addressing the rainfall induced catchment-derived sediment loading at the catchment scale through catchment-sensitive farming measures that identify key sources of sediment input. Addressing such a system re-balance can only be accomplished through holistic and multidisciplinary catchment management plans that deliver rehabilitation at cascading scales from the catchment to mesohabitat level.

Although ecologically desirable, reverting the English landscape into a forested pristine-like condition in pursuit of pre-human levels of sediment input is not tenable. However, identification and pragmatic rehabilitation of system control mechanisms that operate at the catchment-scale can be addressed locally on a reach-by-reach basis through sediment control mechanisms such as interception wetlands, floodplain reconnection and greater stream velocity heterogeneity. Causes of ecological decline cannot be addressed effectively by means of in-stream rehabilitation measures alone. Beechie et al. (2012) argued that in-stream rehabilitation measures do not increase the resilience of salmonid populations. Management efforts should focus at the catchment scale in the first instance (see de Jalón, 1995), developing process-led rehabilitation, addressing land-use and associated sedimentation at the source of the problem. Rehabilitation of natural processes generates ecosystem resilience and subsequently sustainability and the mechanism for natural recovery of biological and physical river functions (Beechie et al., 2012). Time to recovery is the dissuading factor as the perturbation factor slowly works itself through the system. However, complimentary form-led rehabilitation can then be delivered at consecutively smaller spatial scales. Without the appropriate scale of management, the ephemeral nature of rehabilitation gravel will continue to provide a significant barrier to projects of this nature. Planning and developing sustainable solutions under current predictions of environmental and climatic change will remain the greatest challenge to river management (see Macklin and Lewin, 1997).

## 7.2 Emerging questions

The scarcity of studies investigating the function of rehabilitation gravel, particularly in the UK, provides considerable value to this research. As such, the key findings presented here will not only assist stream management and potentially support critical catchment management decisions, specifically those targeting excessive run-off, but will contribute towards river rehabilitation science. As a result, future rehabilitation strategies will be better placed with a greater understanding of the role catchment scale processes contribute to macrohabitat scale integrity, and subsequently *S. trutta* abundance. The installation of rehabilitation gravel should demonstrate greater sensitivity to dominant channel and catchment processes.

Both Pasternack et al. (2004) and Barlaup et al. (2008) have argued for a more heterogeneous mix of rehabilitation gravel sediment sizes. Natural gravels, with a greater sorting coefficient and smaller interstitial void size, maintain greater permeability within the predominantly sand sized (2>$D$≥1 mm) suspended sediment flux observed in chalk streams. Would rehabilitation gravel modelled off of natural gravel composition and structure have increased sustainability for *S. trutta* spawning? Moreover, could a single updated rehabilitation gravel modelled off of natural composition be successfully applied to all rivers, or is it the case that river managers should define a suitable gravel composition based on a localised requirements? Such investigation requires development of suitable guidelines for particular river types that incorporate knowledge of flow hydraulics. Further enquiry in this regard must consider the interdependency of both chemical and biological processes on the physical processes associated with river types.

Moreover, is river rehabilitation by means of gravel introduction a viable management strategy for chalk streams? It is apparent from this research that rehabilitation gravel are somewhat unpredictable as a *S. trutta* spawning habitat and largely unsuitable for persistent recruitment to the *S. trutta* population. As such, rehabilitation by means of introducing a narrow size range of coarse gravel without complementary process-led management can be a high risk management strategy for chalk streams. Comparatively, natural spawning gravels demonstrate greater sustainability and a self regulating nature that give them a key role in maintaining population recruitment. In light of the low abundance of high quality natural spawning gravels in chalk rivers (due to historic channel modification and maintenance), should river managers not give greater scope to rehabilitation of natural gravels by reintroduction of natural processes through, for example, increased abundance of LWD?

### 7.3 Conclusions

A summary of the main conclusions for this study are:

- A historical agricultural economy has dominated the catchment of the River Stiffkey for >200 years. Modification to the river channel in response to agricultural pressures over this time has reduced stream capacity and competence. Deposition of fine sediments in the silt and sand size-range (0.004>$D$>2 mm) has dominated current channel processes. Gravel recruitment was limited and a high in-stream fine sediment supply was maintained. Reductions in total channel length (1.4 km) through straightening of 10 km of channel, a reduction in sinuosity and increased depth have impacted the flow regime of the River Stiffkey.

- Water levels in the River Stiffkey have a reduced response time to rainfall than is typically observed in chalkstreams due to glacial deposits in the upper reaches of the catchment. Sediment-laden run-off, observed during convective rainfall events, mobilises large quantities of sediment from arable fields into the river channel. Due to the characteristic low stream competence the ability of the stream to receive and store excessive catchment-derived sediment loads is high.
- Rehabilitation gravel was installed into modified channel reaches dominated by depositional processes. Rehabilitation gravel reduced the slope angle of the water surface predisposing these sites to further sediment deposition. Moreover, rehabilitation gravels caused water upstream to dam-up behind the gravel exacerbating local depositional processes.
- Installation of rehabilitation gravel did not increase localised water velocity, predisposing these sites to sediment deposition. Given the current hydraulic regime and excessive catchment-derived sediment loadings, rehabilitation gravels cannot sustain their physical integrity and undergo a morphological succession from a very well sorted gravel type towards a poorly sorted deposit. This succession has negative implications for their suitability as a *S. trutta* spawning habitat. It is estimated that the suitability of rehabilitation gravel in the River Stiffkey have a short-term lifespan of <10 years, after which they will be wholly unsuitable for *S. trutta* spawning.
- Overall, ETF survival from rehabilitation gravel was poor. Even after the installation of rehabilitation gravel, only a small number of sites within the study area offered a suitable environment for embryo survival. The variable sediment composition of natural spawning gravels, however, had consistently greater ETF survival than rehabilitation gravel, especially the 2003 rehabilitation gravels. High ETF survival observed within incubation substrate on natural gravel sites were likely due to the characteristic sediment composition and structure; a poorly sorted grain-size distribution and surface armouring. Rehabilitation gravel, however, had little or no stratification and consisted of mostly coarse framework gravels. The larger intragravel void spaces within poorly sorted gravel-dominated rehabilitation gravel have an increased susceptibility to fine sediment accrual. It is likely that this sediment accrual reduced intragravel velocity and permeability. It is assumed that the greater composition of sand within natural gravels reduced interstitial void spaces and subsequent sediment deposition occurred mostly within surface substrate. This would have prevented fine sediments from accumulating

within deeper lying sediments and maintained an area of intragravel permeability beneath surface layers.

- Sediment composition of rehabilitation gravel in the River Stiffkey is sediment-supply controlled and defined by catchment-scale processes. Control of catchment-derived sediment input is therefore key in regulating spawning quality of not only rehabilitation gravel, but also the sustainable management of naturally occurring spawning habitat.
- Low abundance of and poor spatial connectivity between juvenile life-stage specific habitat types created an ecological bottleneck thus further limiting *S. trutta* recruitment potential in the River Stiffkey. Introduction of rehabilitation gravel to the channel in over-deepened reaches did not address key spatial relationship deficiencies between life-stage specific habitat.

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# 9 Appendices

**Appendix 1**.
Totals (g) sampled from rehabilitation gravel site 2003A. Sediment size ranges are in mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core, including sediments *D*>64 mm.

| | Depth (cm) | | **74≥*D*>8** | ***D*=16** | **2>*D*≥1** | ***D*<1** | ***D*<0.004** | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | **$Core_{Tot}$** | **$Site_{Tot}$** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 0-5 | g | 910.09 | 207.76 | 0.21 | 1.86 | 0.03 | 927.56 | 927.37 | 0.02 | | |
| | | % | 98.14 | 22.4 | 0.02 | 0.2 | 0.003 | | | | | |
| | 5-10 | g | 310.87 | 151.36 | 1.9 | 23.55 | 0.74 | 361.65 | 361.6 | 0.01 | | |
| | | % | 85.97 | 41.86 | 0.53 | 6.51 | 0.2 | | | | | |
| | 10-15 | g | 415.93 | 196.33 | 4.42 | 48.24 | 1 | 538.57 | 538.25 | 0.06 | | |
| | | % | 77.27 | 36.48 | 0.82 | 8.96 | 0.19 | | | | | |
| | 15-20 | g | 148.9 | 96.52 | 6.37 | 72.4 | 1.32 | 300.84 | 302.34 | -0.5 | | |
| | | % | 49.25 | 31.92 | 2.11 | 23.95 | 0.44 | | | | | |
| | 20-25 | g | 610.11 | 164.65 | 8.81 | 68.07 | 2.49 | 757.22 | 756.52 | 0.09 | | |
| | | % | 80.65 | 21.76 | 1.16 | 9 | 0.33 | | | | | |
| **Core 1** | 25-30 | g | 216.52 | 25.83 | 8.85 | 85.68 | 3.8 | 364.01 | 363.63 | 0.1 | | |
| | | % | 59.54 | 7.1 | 2.43 | 23.56 | 1.05 | | | | 3249.85 | |
| | 0-5 | g | 626.1 | 90.1 | 0.42 | 2.27 | 0.04 | 630.79 | 630.7 | 0.01 | | |
| | | % | 99.27 | 14.29 | 0.07 | 0.36 | 0.01 | | | | | |
| | 5-10 | g | 378.81 | 79.07 | 0.55 | 7.70 | 0.31 | 398.92 | 398.76 | 0.04 | | |
| | | % | 95 | 19.83 | 0.14 | 1.93 | 0.08 | | | | | |
| | 10-15 | g | 462.04 | 46.27 | 5.01 | 59.76 | 2.78 | 549.25 | 544.68 | 0.83 | | |
| | | % | 84.16 | 8.43 | 0.91 | 10.89 | 0.51 | | | | | |
| | 15-20 | g | 263.77 | 184.57 | 7.22 | 71.31 | 3.84 | 375.08 | 375.04 | 0.01 | | |
| | | % | 70.33 | 49.21 | 1.93 | 19.01 | 1.02 | | | | | |
| | 20-25 | g | 451.92 | 18.45 | 10.14 | 97.80 | 5.07 | 590.73 | 590.23 | 0.08 | | |
| | | % | 76.57 | 3.13 | 1.72 | 16.57 | 0.86 | | | | | |
| **Core 2** | 25-30 | g | 851.85 | 13.7 | 26.83 | 164.36 | 5.45 | 1121.1 | 1120.16 | 0.08 | | |
| | | % | 76.05 | 1.22 | 2.4 | 14.67 | 0.49 | | | | 4815.87 | |
| | 0-5 | g | 373.14 | 177.66 | 1.53 | 10.63 | 0.1 | 398.74 | 399.13 | -0.1 | | |
| | | % | 93.49 | 44.51 | 0.38 | 2.66 | 0.02 | | | | | |
| | 5-10 | g | 190.89 | 190.89 | 7.78 | 79.06 | 1.68 | 347.45 | 347.14 | 0.09 | | |
| | | % | 54.99 | 54.99 | 2.24 | 22.78 | 0.49 | | | | | |
| | 10-15 | g | 109.2 | 109.20 | 7.88 | 80.68 | 1.87 | 321.1 | 320.94 | 0.05 | | |
| | | % | 34.03 | 34.03 | 2.46 | 25.14 | 0.58 | | | | | |
| | 15-20 | g | 164.34 | 164.34 | 10.16 | 100.56 | 3.01 | 379.66 | 379.33 | 0.09 | | |
| | | % | 43.32 | 43.32 | 2.68 | 26.51 | 0.79 | | | | | |
| | 20-25 | g | 276.4 | 276.4 | 2.60 | 36.66 | 1.09 | 380.89 | 380.75 | 0.04 | | |
| | | % | 72.59 | 72.59 | 0.68 | 9.63 | 0.29 | | | | | |
| **Core 3** | 25-30 | g | 236.28 | 170.65 | 4.29 | 61.61 | 2.42 | 337.24 | 337.15 | 0.03 | | |
| | | % | 70.08 | 50.62 | 1.27 | 18.28 | 0.72 | | | | 2165.08 | 10230.8 |

**Appendix 2**.

Totals (g) sampled from substrata at site 2003B. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 691.04 | 454.57 | 2.5 | 21.46 | 0.43 | 817.07 | 817.32 | -0.03 | | |
| | | % | 84.55 | 55.62 | 0.31 | 2.63 | 0.05 | | | | | |
| | 5-10 | g | 383.07 | 159.51 | 7.27 | 84.66 | 2.43 | 566.59 | 565.77 | 0.14 | | |
| | | % | 67.71 | 28.19 | 1.29 | 14.96 | 0.43 | | | | | |
| | 10-15 | g | 421.16 | 156.47 | 16.52 | 185.65 | 8.6 | 783.39 | 783.11 | 0.04 | | |
| | | % | 53.78 | 19.98 | 2.11 | 23.71 | 1.1 | | | | | |
| | 15-20 | g | 948.95 | 53.55 | 23.08 | 295.39 | 18.65 | 1373.95 | 1375.24 | -0.09 | | |
| | | % | 69 | 3.89 | 1.68 | 21.48 | 1.36 | | | | | |
| | 20-25 | g | 118.42 | 37.06 | 11.06 | 128.27 | 5.06 | 295.82 | 295.62 | 0.07 | | |
| | | % | 40.06 | 12.54 | 3.74 | 43.39 | 1.71 | | | | | |
| | 25-30 | g | 77.73 | 77.73 | 8.43 | 115.27 | 5.29 | 229.09 | 228.87 | 0.1 | | |
| | | % | 33.96 | 33.96 | 3.68 | 50.36 | 2.31 | | | | 8415.91 | |
| Core 2 | 0-5 | g | 643.58 | 458.39 | 2.46 | 19.99 | 0.16 | 715.16 | 715.13 | 0.004 | | |
| | | % | 89.99 | 64.1 | 0.34 | 2.8 | 0.02 | | | | | |
| | 5-10 | g | 498.59 | 324.75 | 5.89 | 40.93 | 0.34 | 625.79 | 625.36 | 0.07 | | |
| | | % | 79.73 | 51.93 | 0.94 | 6.55 | 0.05 | | | | | |
| | 10-15 | g | 455.59 | 284.88 | 12.61 | 80.47 | 1.06 | 650.05 | 649.55 | 0.08 | | |
| | | % | 70.14 | 43.86 | 1.94 | 12.39 | 0.16 | | | | | |
| | 15-20 | g | 196.22 | 196.22 | 4.36 | 28.75 | 0.19 | 257.85 | 257.58 | 0.1 | | |
| | | % | 76.18 | 76.18 | 1.69 | 11.16 | 0.07 | | | | | |
| | 20-25 | % | - | - | - | - | - | - | - | - | | |
| | | g | - | - | - | - | - | | | | | |
| | 25-30 | % | - | - | - | - | - | - | - | - | | |
| | | g | - | - | - | - | - | | | | 2248.85 | |
| Core 3 | 0-5 | g | 822.63 | 262.92 | 1.25 | 3.34 | 0.02 | 869.28 | 869.55 | -0.03 | | |
| | | % | 94.6 | 30.24 | 0.14 | 0.38 | 0.003 | | | | | |
| | 5-10 | g | 332.65 | 260.77 | 2.06 | 15.02 | 0.24 | 388.48 | 388.64 | -0.04 | | |
| | | % | 85.59 | 67.1 | 0.53 | 3.86 | 0.06 | | | | | |
| | 10-15 | g | 213.86 | 213.86 | 6.14 | 41.63 | 1.2 | 386.18 | 386.21 | -0.01 | | |
| | | % | 55.37 | 55.37 | 1.59 | 10.78 | 0.31 | | | | | |
| | 15-20 | g | 229.52 | 195.65 | 11.89 | 71.05 | 2.64 | 445.01 | 445.08 | -0.02 | | |
| | | % | 51.57 | 43.96 | 2.67 | 15.96 | 0.59 | | | | | |
| | 20-25 | g | 694.96 | 72.97 | 8.61 | 64.39 | 1.91 | 840.96 | 841.30 | -0.04 | | |
| | | % | 82.61 | 8.67 | 1.02 | 7.65 | 0.23 | | | | | |
| | 25-30 | g | 47.95 | 47.95 | 12.29 | 96.28 | 6.09 | 240.76 | 240.63 | 0.05 | | |
| | | % | 19.93 | 19.93 | 5.11 | 40.01 | 2.53 | | | | 5370.67 | 16035.43 |

**Appendix 3**.

Totals (g) sampled from substrata at site 2003C. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 301.14 | 189.17 | 8.31 | 61.48 | 0.84 | 395.65 | 390.9 | 1.2 | | |
| | | % | 77.04 | 48.39 | 2.13 | 15.73 | 0.21 | | | | | |
| | 5-10 | g | 621.58 | 389.42 | 31.93 | 194.4 | 5 | 1003.15 | 997.49 | 0.56 | | |
| | | % | 62.31 | 39.04 | 3.2 | 19.49 | 0.5 | | | | | |
| | 10-15 | g | 687.78 | 528.3 | 32.77 | 204.06 | 5.11 | 1101.07 | 1094.45 | 0.6 | | |
| | | % | 62.84 | 48.27 | 2.99 | 18.64 | 0.47 | | | | | |
| | 15-20 | g | 501.95 | 394.98 | 26.05 | 167.68 | 5.23 | 808.46 | 802.44 | 0.74 | | |
| | | % | 62.55 | 49.22 | 3.25 | 20.9 | 0.65 | | | | | |
| | 20-25 | g | 398.32 | 268.35 | 18.9 | 148.93 | 5.14 | 642.89 | 637.25 | 0.88 | | |
| | | % | 62.51 | 42.11 | 2.97 | 23.37 | 0.81 | | | | | |
| | 25-30 | g | 31.93 | 31.93 | 2.7 | 33.35 | 2.26 | 79.18 | 78.75 | 0.54 | | |
| | | % | 40.55 | 40.55 | 3.42 | 42.35 | 2.87 | | | | 10465.15 | |
| Core 2 | 0-5 | g | 325.86 | 235.41 | 2.83 | 17.05 | 0.69 | 390.31 | 385.87 | 1.14 | | |
| | | % | 84.45 | 61.01 | 0.73 | 4.42 | 0.18 | | | | | |
| | 5-10 | g | 127.92 | 127.92 | 14.16 | 90.2 | 3.79 | 352.92 | 350.4 | 0.71 | | |
| | | % | 36.51 | 36.51 | 4.04 | 25.74 | 1.08 | | | | | |
| | 10-15 | g | 263.45 | 61.48 | 12.69 | 81.95 | 3.21 | 414.84 | 410.61 | 1.02 | | |
| | | % | 64.16 | 14.97 | 3.09 | 19.96 | 0.78 | | | | | |
| | 15-20 | g | 133.41 | 82.46 | 15.96 | 113.69 | 7.1 | 361.82 | 355.09 | 1.86 | | |
| | | % | 37.57 | 23.22 | 4.49 | 32.02 | 2 | | | | | |
| | 20-25 | g | 6.67 | 6.67 | 13.5 | 102.21 | 6.34 | 195.04 | 186.98 | 4.13 | | |
| | | % | 3.57 | 3.57 | 7.22 | 54.66 | 3.39 | | | | | |
| | 25-30 | g | 111.21 | 111.21 | 12.92 | 108.75 | 7.55 | 305.5 | 295.96 | 3.12 | | |
| | | % | 37.58 | 37.58 | 4.37 | 36.74 | 2.55 | | | | 7160.43 | |
| Core 3 | 0-5 | g | 305.71 | 46.56 | 2.82 | 20.38 | 0.37 | 346.22 | 341.39 | 1.4 | | |
| | | % | 89.55 | 13.64 | 0.83 | 5.97 | 0.11 | | | | | |
| | 5-10 | g | 114.76 | 59.84 | 5.33 | 30 | 0.43 | 167.52 | 163.01 | 2.69 | | |
| | | % | 70.4 | 36.71 | 3.27 | 18.41 | 0.27 | | | | | |
| | 10-15 | g | 71.93 | 71.93 | 6.16 | 46.27 | 0.78 | 157.35 | 157.18 | 0.11 | | |
| | | % | 45.76 | 45.76 | 3.92 | 29.44 | 0.49 | | | | | |
| | 15-20 | g | 48.4 | 48.4 | 12.38 | 90.45 | 5.79 | 197.56 | 196.72 | 0.43 | | |
| | | % | 24.6 | 24.6 | 6.29 | 45.98 | 2.94 | | | | | |
| | 20-25 | g | 11.17 | 11.17 | 8.3 | 68.52 | 4.95 | 112.98 | 112.34 | 0.57 | | |
| | | % | 9.94 | 9.94 | 7.39 | 60.99 | 4.4 | | | | | |
| | 25-30 | g | 0 | 0 | 8.88 | 80.41 | 6.24 | 126.83 | 125.94 | 0.7 | | |
| | | % | 0 | 0 | 7.05 | 63.85 | 4.96 | | | | 3158.46 | 20784.04 |

**Appendix 4**.

Totals (g) sampled from substrata at site 2009A. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments $D$>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 358.34 | 263.24 | 0.51 | 0.61 | 0.002 | 366.83 | 366.75 | 0.02 | | |
| | | % | 97.71 | 71.78 | 0.14 | 0.17 | 0.001 | | | | | |
| | 5-10 | g | 347.58 | 144.84 | 0.34 | 1.15 | 0.01 | 370.7 | 370.54 | 0.04 | | |
| | | % | 93.80 | 39.09 | 0.09 | 0.31 | 0.001 | | | | | |
| | 10-15 | g | 274.16 | 183.2 | 0.53 | 4.75 | 0.05 | 319.86 | 319.58 | 0.09 | | |
| | | % | 85.79 | 57.33 | 0.17 | 1.49 | 0.02 | | | | | |
| | 15-20 | g | 159.02 | 159.02 | 0.85 | 13.53 | 0.26 | 229.13 | 229 | 0.06 | | |
| | | % | 69.44 | 69.44 | 0.37 | 5.91 | 0.11 | | | | | |
| | 20-25 | g | 118.14 | 118.14 | 3.54 | 38.45 | 0.53 | 306.9 | 306.47 | 0.14 | | |
| | | % | 38.55 | 38.55 | 1.16 | 12.55 | 0.17 | | | | | |
| | 25-30 | g | 123.21 | 123.21 | 6.18 | 44.48 | 0.87 | 252.05 | 251.71 | 0.13 | | |
| | | % | 48.95 | 48.95 | 2.45 | 17.67 | 0.35 | | | | 1845.47 | |
| Core 2 | 0-5 | g | 561.01 | 308.47 | 2.9 | 15.34 | 0.36 | 827.03 | 826.26 | 0.09 | | |
| | | % | 67.9 | 37.33 | 0.35 | 1.86 | 0.04 | | | | | |
| | 5-10 | g | 539.29 | 49.87 | 3.49 | 21.18 | 0.21 | 667.51 | 667.11 | 0.06 | | |
| | | % | 80.84 | 7.48 | 0.52 | 3.18 | 0.03 | | | | | |
| | 10-15 | g | 763.53 | 209.76 | 5.27 | 38.15 | 1.3 | 925.77 | 924.99 | 0.08 | | |
| | | % | 82.54 | 22.68 | 0.57 | 4.12 | 0.14 | | | | | |
| | 15-20 | g | 74.95 | 74.95 | 3.69 | 33.91 | 1.61 | 184.58 | 184.38 | 0.11 | | |
| | | % | 40.65 | 40.65 | 2 | 18.39 | 0.88 | | | | | |
| | 20-25 | g | 617.08 | 164.05 | 9.05 | 67.30 | 1.21 | 793.31 | 792.54 | 0.1 | | |
| | | % | 77.86 | 20.7 | 1.14 | 8.49 | 0.15 | | | | | |
| | 25-30 | g | 268.16 | 169.30 | 15.28 | 99.02 | 2.42 | 614.51 | 612.9 | 0.26 | | |
| | | % | 43.75 | 27.62 | 2.49 | 16.16 | 0.39 | | | | 12787.71 | |
| Core 3 | 0-5 | g | 600.84 | 402.62 | 0.90 | 3.75 | 0.03 | 616.75 | 616.57 | 0.03 | | |
| | | % | 97.45 | 65.30 | 0.15 | 0.61 | 0.005 | | | | | |
| | 5-10 | g | 272.41 | 128.22 | 3.19 | 22.12 | 0.63 | 322.75 | 322.41 | 0.11 | | |
| | | % | 84.49 | 39.77 | 0.99 | 6.86 | 0.19 | | | | | |
| | 10-15 | g | 203.42 | 203.42 | 7.40 | 46.61 | 1.06 | 360.56 | 360.11 | 0.12 | | |
| | | % | 56.49 | 56.49 | 2.06 | 12.94 | 0.29 | | | | | |
| | 15-20 | g | 203.75 | 203.75 | 5.02 | 41.42 | 0.89 | 378.14 | 378.14 | 0 | | |
| | | % | 53.88 | 53.88 | 1.33 | 10.95 | 0.24 | | | | | |
| | 20-25 | g | 352.52 | 179.00 | 4.66 | 34.52 | 0.94 | 491.2 | 490.87 | 0.07 | | |
| | | % | 71.82 | 36.47 | 0.95 | 7.03 | 0.19 | | | | | |
| | 25-30 | g | 298.33 | 212.31 | 2.70 | 34.99 | 1.42 | 419.25 | 418.75 | 0.12 | | |
| | | % | 71.24 | 50.7 | 0.65 | 8.36 | 0.34 | | | | 2588.65 | 17221.83 |

**Appendix 5**.

Totals (g) sampled from substrata at site 2009D. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 675.95 | 320.79 | 0.3 | 2.16 | 0.02 | 696.35 | 692.34 | 0.58 | | |
| | | % | 97.63 | 46.33 | 0.04 | 0.31 | 0.002 | | | | | |
| | 5-10 | g | 450.47 | 312.59 | 0.45 | 6.73 | 0.06 | 492.53 | 488.78 | 0.76 | | |
| | | % | 92.16 | 63.95 | 0.09 | 1.38 | 0.01 | | | | | |
| | 10-15 | g | 350.28 | 266.29 | 2.22 | 27.61 | 0.92 | 436.85 | 433.62 | 0.74 | | |
| | | % | 80.78 | 61.41 | 0.51 | 6.37 | 0.21 | | | | | |
| | 15-20 | g | 264.8 | 212.08 | 8.03 | 103.3 | 2.99 | 473.24 | 474.08 | -0.18 | | |
| | | % | 55.86 | 44.74 | 1.69 | 21.79 | 0.63 | | | | | |
| | 20-25 | g | 204.18 | 204.18 | 5.99 | 66.23 | 1.76 | 368.62 | 371.2 | -0.7 | | |
| | | % | 55.01 | 55.01 | 1.61 | 17.84 | 0.47 | | | | | |
| | 25-30 | g | 57.89 | 57.89 | 5.51 | 84.95 | 3.24 | 205.47 | 206.72 | -0.61 | | |
| | | % | 28 | 28 | 2.67 | 41.09 | 1.57 | | | | 6529.16 | |
| Core 2 | 0-5 | g | 695.42 | 287.91 | 0.65 | 9.59 | 0.37 | 793.42 | 789.47 | 0.5 | | |
| | | % | 88.09 | 36.47 | 0.08 | 1.21 | 0.05 | | | | | |
| | 5-10 | g | 292.9 | 177.39 | 2.71 | 51 | 1.73 | 437.14 | 433.81 | 0.76 | | |
| | | % | 67.52 | 40.89 | 0.62 | 11.76 | 0.4 | | | | | |
| | 10-15 | g | 218.71 | 218.71 | 2.44 | 53.03 | 1.84 | 400.87 | 401.75 | -0.22 | | |
| | | % | 54.44 | 54.44 | 0.61 | 13.20 | 0.46 | | | | | |
| | 15-20 | g | 199.06 | 121.02 | 4.63 | 66.25 | 2.3 | 387.66 | 384.04 | 0.93 | | |
| | | % | 51.83 | 31.51 | 1.2 | 17.25 | 0.6 | | | | | |
| | 20-25 | g | 176.6 | 124.64 | 3.76 | 41.02 | 1.47 | 303.91 | 300.74 | 1.04 | | |
| | | % | 58.72 | 41.44 | 1.25 | 13.64 | 0.49 | | | | | |
| | 25-30 | g | 462.47 | 181.7 | 6.63 | 65.93 | 2.21 | 683.73 | 680.55 | 0.47 | | |
| | | % | 67.96 | 26.7 | 0.97 | 9.69 | 0.32 | | | | 3006.73 | |
| Core 3 | 0-5 | g | 578.46 | 251.51 | 0.98 | 5.46 | 0.1 | 618.87 | 614.26 | 0.74 | | |
| | | % | 94.17 | 40.95 | 0.16 | 0.89 | 0.02 | | | | | |
| | 5-10 | g | 387.82 | 205.82 | 7.46 | 39.38 | 0.75 | 477.75 | 472.89 | 1.02 | | |
| | | % | 82.01 | 43.52 | 1.58 | 8.33 | 0.16 | | | | | |
| | 10-15 | g | 191.83 | 138.11 | 17.78 | 66.46 | 0.85 | 366.53 | 360.48 | 1.65 | | |
| | | % | 53.22 | 38.31 | 4.93 | 18.44 | 0.24 | | | | | |
| | 15-20 | g | 187.71 | 140.26 | 18.54 | 95.02 | 1.97 | 432.88 | 427.73 | 1.19 | | |
| | | % | 43.89 | 32.79 | 4.33 | 22.22 | 0.46 | | | | | |
| | 20-25 | g | 51.86 | 10.29 | 19.4 | 173.30 | 2.26 | 301.18 | 295.86 | 1.77 | | |
| | | % | 17.53 | 3.48 | 6.56 | 58.57 | 0.76 | | | | | |
| | 25-30 | g | 0 | 0 | 20.41 | 240.77 | 2.7 | 271.44 | 267.84 | 1.33 | | |
| | | % | 0 | 0 | 7.62 | 89.89 | 1.01 | | | | 2468.65 | 12004.54 |

**Appendix 6**.

Totals (g) sampled from substrata at site 2009F. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| Core | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 519.13 | 286.38 | 5.93 | 31.77 | 0.65 | 572.1 | 572 | 0.02 | | |
| | | % | 90.76 | 50.07 | 1.04 | 5.55 | 0.11 | | | | | |
| | 5-10 | g | 339.9 | 277.62 | 12.95 | 78.89 | 1.49 | 519.74 | 519.8 | -0.01 | | |
| | | % | 65.39 | 53.41 | 2.49 | 15.18 | 0.29 | | | | | |
| | 10-15 | g | 129.89 | 129.89 | 4.74 | 20.7 | 0.2 | 184.73 | 184.24 | 0.27 | | |
| | | % | 70.5 | 70.5 | 2.57 | 11.23 | 0.11 | | | | | |
| | 15-20 | g | 199.44 | 199.44 | 11.21 | 51.84 | 0.78 | 370.9 | 370.74 | 0.04 | | |
| | | % | 53.8 | 53.8 | 3.02 | 13.98 | 0.21 | | | | | |
| | 20-25 | g | 97.04 | 97.04 | 7.31 | 36.44 | 0.95 | 242.55 | 242.2 | 0.14 | | |
| | | % | 40.07 | 40.07 | 3.02 | 15.05 | 0.39 | | | | | |
| | 25-30 | g | 98.36 | 98.36 | 5.68 | 28.2 | 0.32 | 213.25 | 212.95 | 0.14 | | |
| | | % | 46.19 | 46.19 | 2.67 | 13.24 | 0.15 | | | | 2103.27 | |
| Core 2 | 0-5 | g | 726.35 | 357.99 | 0.84 | 16.52 | 0.58 | 773.14 | 772.9 | 0.03 | | |
| | | % | 93.98 | 46.32 | 0.11 | 2.14 | 0.08 | | | | | |
| | 5-10 | g | 419.05 | 265.73 | 7.02 | 68.26 | 1.21 | 538.11 | 537.8 | 0.06 | | |
| | | % | 77.92 | 49.41 | 1.31 | 12.69 | 0.23 | | | | | |
| | 10-15 | g | 427.38 | 199.38 | 10.21 | 85.57 | 1.79 | 648.18 | 647.58 | 0.09 | | |
| | | % | 66 | 30.79 | 1.58 | 13.21 | 0.28 | | | | | |
| | 15-20 | g | 250.17 | 108.65 | 6.69 | 49.44 | 1.05 | 385.02 | 384.79 | 0.06 | | |
| | | % | 65.01 | 28.24 | 1.74 | 12.85 | 0.27 | | | | | |
| | 20-25 | g | 274.89 | 274.89 | 4.92 | 41.58 | 1.42 | 433.56 | 433.35 | 0.05 | | |
| | | % | 63.43 | 63.43 | 1.14 | 9.59 | 0.33 | | | | | |
| | 25-30 | g | 373.02 | 152.98 | 5.64 | 40.95 | 1.02 | 541.56 | 541.48 | 0.01 | | |
| | | % | 68.89 | 28.25 | 1.04 | 7.56 | 0.19 | | | | 3319.57 | |
| Core 3 | 0-5 | g | 439.93 | 258.68 | 1.16 | 5.35 | 0.08 | 467.54 | 467.78 | -0.05 | | |
| | | % | 94.05 | 55.3 | 0.25 | 1.14 | 0.02 | | | | | |
| | 5-10 | g | 192.59 | 192.59 | 15.19 | 54.55 | 0.7 | 344.38 | 344.38 | 0 | | |
| | | % | 55.92 | 55.92 | 4.41 | 15.84 | 0.2 | | | | | |
| | 10-15 | g | 337.95 | 225.03 | 16.59 | 55.35 | 0.82 | 471.57 | 471.36 | 0.04 | | |
| | | % | 71.7 | 47.74 | 3.52 | 11.74 | 0.17 | | | | | |
| | 15-20 | g | 187.71 | 187.71 | 17.11 | 58.51 | 0.92 | 388.16 | 388.18 | -0.01 | | |
| | | % | 48.36 | 48.36 | 4.41 | 15.07 | 0.24 | | | | | |
| | 20-25 | g | 182.38 | 182.38 | 11.67 | 39.06 | 0.64 | 368.31 | 368.31 | 0 | | |
| | | % | 49.52 | 49.52 | 3.17 | 10.61 | 0.17 | | | | | |
| | 25-30 | g | 246.09 | 188.97 | 3.71 | 11.35 | 0.15 | 326.46 | 326.94 | -0.15 | | |
| | | % | 75.27 | 57.8 | 1.14 | 3.47 | 0.05 | | | | 2366.42 | 7789.26 |

**Appendix 7**.

Totals (g) sampled from substrata at site 2009J Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 497.88 | 69.88 | 0.24 | 27.81 | 0.7 | 557.53 | 551.37 | 1.1 | | |
| | | % | 90.3 | 12.67 | 0.04 | 5.04 | 0.13 | | | | | |
| | 5-10 | g | 501.64 | 214.17 | 0.42 | 34.16 | 1.12 | 598.39 | 593.05 | 0.89 | | |
| | | % | 84.59 | 36.11 | 0.07 | 5.76 | 0.19 | | | | | |
| | 10-15 | g | 356.7 | 173.63 | 0.96 | 26.16 | 1.21 | 505.42 | 500.3 | 1.01 | | |
| | | % | 71.3 | 34.71 | 0.19 | 5.23 | 0.24 | | | | | |
| | 15-20 | g | 396.12 | 278.84 | 0.94 | 32.09 | 2.02 | 600.53 | 595.28 | 0.87 | | |
| | | % | 66.54 | 46.84 | 0.16 | 5.39 | 0.34 | | | | | |
| | 20-25 | g | 348.71 | 125.31 | 0.97 | 22.32 | 1.15 | 500.53 | 495.29 | 1.05 | | |
| | | % | 70.41 | 25.3 | 0.2 | 4.51 | 0.23 | | | | | |
| | 25-30 | g | 257.4 | 257.4 | 0.67 | 17.2 | 1.08 | 372.83 | 372.33 | 0.13 | | |
| | | % | 69.13 | 69.13 | 0.18 | 4.62 | 0.29 | | | | 3135.23 | |
| Core 2 | 0-5 | g | 225.3 | 22.28 | 0.01 | 3.2 | 0.11 | 233.36 | 228.55 | 2.06 | | |
| | | % | 98.58 | 9.75 | 0.004 | 1.4 | 0.05 | | | | | |
| | 5-10 | g | 646.96 | 205.7 | 0.02 | 9.85 | 0.49 | 694.74 | 688.89 | 0.84 | | |
| | | % | 93.91 | 29.86 | 0.003 | 1.43 | 0.07 | | | | | |
| | 10-15 | g | 379.52 | 110.6 | 0.04 | 13.91 | 0.54 | 491.72 | 486.31 | 1.1 | | |
| | | % | 78.04 | 22.74 | 0.01 | 2.86 | 0.11 | | | | | |
| | 15-20 | g | 316.36 | 77.56 | 0.09 | 32.7 | 1.23 | 464.08 | 458.29 | 1.25 | | |
| | | % | 69.03 | 16.92 | 0.02 | 7.14 | 0.27 | | | | | |
| | 20-25 | g | 522.23 | 158.07 | 0.48 | 50.46 | 2.28 | 786.17 | 779.46 | 0.85 | | |
| | | % | 67 | 20.28 | 0.06 | 6.47 | 0.29 | | | | | |
| | 25-30 | g | 533.78 | 98.83 | 0.45 | 50.98 | 2.36 | 690.71 | 683.33 | 1.07 | | |
| | | % | 78.11 | 14.46 | 0.07 | 7.46 | 0.35 | | | | 6285.78 | |
| Core 3 | 0-5 | g | 1015.47 | 184.44 | 0.32 | 13.8 | 0.64 | 1099.19 | 1092.2 | 0.64 | | |
| | | % | 92.97 | 16.89 | 0.03 | 1.26 | 0.06 | | | | | |
| | 5-10 | g | 253 | 201.2 | 1.97 | 34.49 | 1.53 | 448.65 | 442.89 | 1.28 | | |
| | | % | 57.12 | 45.43 | 0.44 | 7.79 | 0.35 | | | | | |
| | 10-15 | g | 536.43 | 106.19 | 2.28 | 25.16 | 1.27 | 677.29 | 671.52 | 0.85 | | |
| | | % | 79.88 | 15.81 | 0.34 | 3.75 | 0.19 | | | | | |
| | 15-20 | g | 155.65 | 155.65 | 4.27 | 32.19 | 1.34 | 402.54 | 401.48 | 0.26 | | |
| | | % | 38.77 | 38.77 | 1.06 | 8.02 | 0.33 | | | | | |
| | 20-25 | g | 110.55 | 110.55 | 8.03 | 41.13 | 1.97 | 461.71 | 459.67 | 0.44 | | |
| | | % | 24.05 | 24.05 | 1.75 | 8.95 | 0.43 | | | | | |
| | 25-30 | g | 79.94 | 8.28 | 7.67 | 75.14 | 6.07 | 414.77 | 406.65 | 1.96 | | |
| | | % | 19.66 | 2.04 | 1.89 | 18.48 | 1.49 | | | | 10984 | 20405.01 |

**Appendix 8**.

Totals (g) sampled from substrata at site Fort. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments $D$>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 233.99 | 199.76 | 3.45 | 20.76 | 1.5 | - | 321.09 | - | | |
| | | % | 73.27 | 62.55 | 1.08 | 6.5 | 0.47 | | | | | |
| | 5-10 | g | 340.61 | 70.16 | 28.09 | 184.97 | 12.1 | - | 732.34 | - | | |
| | | % | 46.92 | 9.66 | 3.87 | 25.48 | 1.67 | | | | | |
| | 10-15 | g | 141.45 | 90.42 | 33.3 | 173.41 | 22.33 | - | 481.14 | - | | |
| | | % | 30.19 | 19.3 | 7.11 | 37.01 | 4.77 | | | | | |
| | 15-20 | g | 235.6 | 94.09 | 25.19 | 166.37 | 53.81 | - | 527.17 | - | | |
| | | % | 44.42 | 17.74 | 4.75 | 31.37 | 10.15 | | | | | |
| | 20-25 | g | 138.55 | 55.18 | 13.62 | 113.02 | 37.99 | - | 378.62 | - | | |
| | | % | 38.09 | 15.17 | 3.74 | 31.07 | 10.44 | | | | | |
| | 25-30 | g | 22.47 | 22.47 | 16.42 | 83.89 | 30.03 | - | 243.29 | - | 3308.64 | |
| | | % | 9.61 | 9.61 | 7.02 | 35.86 | 12.84 | | | | | |
| Core 2 | 0-5 | g | 341.94 | 168.19 | 10.25 | 22.24 | 3.28 | 494.21 | 492.02 | 0.44 | | |
| | | % | 69.5 | 34.18 | 2.08 | 4.52 | 0.67 | | | | | |
| | 5-10 | g | 48.67 | 48.67 | 27.64 | 83.44 | 8.57 | 314.79 | 318.49 | -1.18 | | |
| | | % | 15.28 | 15.28 | 8.68 | 26.2 | 2.69 | | | | | |
| | 10-15 | g | 77.03 | 44.83 | 43.41 | 180.47 | 15.72 | 497.79 | 496.56 | 0.25 | | |
| | | % | 15.51 | 9.03 | 8.74 | 36.34 | 3.17 | | | | | |
| | 15-20 | g | 92.75 | 63.01 | 41.82 | 187.96 | 31.53 | 465.23 | 469.51 | -0.92 | | |
| | | % | 19.75 | 13.42 | 8.91 | 40.03 | 6.71 | | | | | |
| | 20-25 | g | 254.32 | 114.82 | 45.56 | 216.76 | 16.87 | 849.21 | 725.3 | 14.59 | | |
| | | % | 35.06 | 15.83 | 6.28 | 29.89 | 2.33 | | | | | |
| | 25-30 | g | 219.09 | 101.38 | 23.22 | 198.11 | 71.29 | 586.11 | 584.6 | 0.26 | 4932.34 | |
| | | % | 37.48 | 17.34 | 3.97 | 33.89 | 12.2 | | | | | |
| Core 3 | 0-5 | g | 149.36 | 118.41 | 7.2 | 107.5 | 25.6 | - | 383.14 | - | | |
| | | % | 38.98 | 30.91 | 1.88 | 28.06 | 6.68 | | | | | |
| | 5-10 | g | 53.53 | 53.53 | 17.18 | 189.66 | 63.01 | - | 382.44 | - | | |
| | | % | 14.00 | 14 | 4.49 | 49.59 | 16.48 | | | | | |
| | 10-15 | g | 60.96 | 60.96 | 11.7 | 137.21 | 59.5 | - | 306.19 | - | | |
| | | % | 19.91 | 19.91 | 3.82 | 44.81 | 19.43 | | | | | |
| | 15-20 | g | 84.91 | 84.91 | 9.94 | 110.36 | 45.36 | - | 284.16 | - | | |
| | | % | 29.88 | 29.88 | 3.5 | 38.84 | 15.96 | | | | | |
| | 20-25 | g | 111.36 | 33.56 | 11.31 | 124.96 | 54.16 | 336.77 | 330.14 | 1.97 | | |
| | | % | 33.73 | 10.17 | 3.43 | 37.85 | 16.4 | | | | | |
| | 25-30 | g | 48.47 | 48.47 | 15.13 | 151.07 | 66.39 | 320.34 | 322.26 | -0.6 | 2008.32 | 10249.31 |
| | | % | 15.04 | 15.04 | 4.69 | 46.88 | 20.6 | | | | | |

**Appendix 9**.

Totals (g) sampled from substrata at site Water Hall. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 122.14 | 122.14 | 12.93 | 53.23 | 1.07 | 307.8 | 307.41 | 0.13 | | |
| | | % | 39.73 | 39.73 | 4.21 | 17.32 | 0.35 | | | | | |
| | 5-10 | g | 84.97 | 84.97 | 25.2 | 120.66 | 4.54 | 453.2 | 456.13 | -0.65 | | |
| | | % | 18.63 | 18.63 | 5.52 | 26.45 | 1 | | | | | |
| | 10-15 | g | 108.61 | 108.61 | 34.29 | 185.25 | 7.18 | 607.5 | 606.46 | 0.17 | | |
| | | % | 17.91 | 17.91 | 5.65 | 30.55 | 1.18 | | | | | |
| | 15-20 | g | 340.16 | 117.12 | 31.09 | 218.34 | 11.84 | 847.63 | 844.01 | 0.43 | | |
| | | % | 40.3 | 13.88 | 3.68 | 25.87 | 1.4 | | | | | |
| | 20-25 | g | 162.01 | 132.81 | 34.71 | 225.32 | 12.9 | 656.91 | 651.41 | 0.84 | | |
| | | % | 24.87 | 20.39 | 5.33 | 34.59 | 1.98 | | | | | |
| | 25-30 | g | 34.26 | 34.26 | 22.93 | 178.66 | 14.52 | 421.6 | 424.65 | -0.72 | | |
| | | % | 8.07 | 8.07 | 5.4 | 42.07 | 3.42 | | | | 3294.64 | |
| Core 2 | 0-5 | g | 326.51 | 270.29 | 8.72 | 64.43 | 1.3 | 525.09 | 520.02 | 0.97 | | |
| | | % | 62.79 | 51.98 | 1.68 | 12.39 | 0.25 | | | | | |
| | 5-10 | g | 302.44 | 214.33 | 28.3 | 203.85 | 4.6 | 722.26 | 716.71 | 0.77 | | |
| | | % | 42.2 | 29.9 | 3.95 | 28.44 | 0.64 | | | | | |
| | 10-15 | g | 167.87 | 136.99 | 22.98 | 175.02 | 9.54 | 565.07 | 560.32 | 0.84 | | |
| | | % | 29.96 | 24.45 | 4.10 | 31.24 | 1.7 | | | | | |
| | 15-20 | g | 192.75 | 147.71 | 39.51 | 245.36 | 35.59 | 830.76 | 839.87 | -1.1 | | |
| | | % | 22.95 | 17.59 | 4.7 | 29.21 | 4.24 | | | | | |
| | 20-25 | g | 456.29 | 254.19 | 41.31 | 271.67 | 23.54 | 1156.81 | 1152.06 | 0.41 | | |
| | | % | 39.61 | 22.06 | 3.59 | 23.58 | 2.04 | | | | | |
| | 25-30 | g | 339.59 | 211.83 | 42.96 | 319.33 | 16.53 | 1049.35 | 1039.78 | 0.91 | | |
| | | % | 32.66 | 20.37 | 4.13 | 30.71 | 1.59 | | | | 5281.14 | |
| Core 3 | 0-5 | g | 255.54 | 197.83 | 1.31 | 5.53 | 0.17 | 363.48 | 358.9 | 1.26 | | |
| | | % | 71.2 | 55.12 | 0.36 | 1.54 | 0.05 | | | | | |
| | 5-10 | g | 171.11 | 171.11 | 16.35 | 83.19 | 1.91 | 481.4 | 480.56 | 0.17 | | |
| | | % | 35.61 | 35.61 | 3.4 | 17.31 | 0.4 | | | | | |
| | 10-15 | g | 264.3 | 52.46 | 32.33 | 183.86 | 3.81 | 723.51 | 717.96 | 0.77 | | |
| | | % | 36.81 | 7.31 | 4.5 | 25.61 | 0.53 | | | | | |
| | 15-20 | g | 125.8 | 125.8 | 26.42 | 177.61 | 5.13 | 466.75 | 465.72 | 0.22 | | |
| | | % | 27.01 | 27.01 | 5.67 | 38.14 | 1.1 | | | | | |
| | 20-25 | g | 319.99 | 166.89 | 30.3 | 204.55 | 9.27 | 789.58 | 784.22 | 0.68 | | |
| | | % | 40.8 | 21.28 | 3.86 | 26.08 | 1.18 | | | | | |
| | 25-30 | g | 181.03 | 181.03 | 24.18 | 146.11 | 7.23 | 611.2 | 610.52 | 0.11 | | |
| | | % | 29.65 | 29.65 | 3.96 | 23.93 | 1.18 | | | | 3435.92 | 12011.7 |

**Appendix 10**.

Totals (g) sampled from substrata at site Whey Curd. Sediment size ranges are in mm. Total core weight (Core $_{tot}$) includes sediments *D*>64 mm. $g_{tot}$ is the total weight (g) of each 5 cm stratification prior to analysis, $g_{Cum}$ is the cumulative weight (g) of each sediment size fraction post-analysis, $E_{(\%)}$ is the error (%) between $g_{tot}$ and $g_{Cum}$, and $Core_{Tot}$ is the total weight (g) of the core.

| | Depth (cm) | | 74≥*D*>8 | *D*=16 | 2>*D*≥1 | *D*<1 | *D*<0.004 | $g_{Tot}$ | $g_{Cum}$ | $g_{Err(\%)}$ | $Core_{Tot}$ | $Site_{Tot}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Core 1 | 0-5 | g | 267.56 | 221.78 | 19.61 | 134.5 | 3.37 | 615.81 | 620.51 | -0.76 | | |
| | | % | 43.12 | 35.74 | 3.16 | 21.68 | 0.54 | | | | | |
| | 5-10 | g | 920.91 | 68.07 | 13.29 | 90.66 | 1.85 | 1149.12 | 1148.37 | 0.07 | | |
| | | % | 80.19 | 5.93 | 1.16 | 7.9 | 0.16 | | | | | |
| | 10-15 | g | 89.95 | 89.95 | 20.92 | 118.08 | 6.53 | 365.79 | 370.81 | -1.37 | | |
| | | % | 24.26 | 24.26 | 5.64 | 31.84 | 1.76 | | | | | |
| | 15-20 | g | 50.42 | 50.42 | 15.57 | 153.16 | 11.84 | 306.34 | 307.41 | -0.35 | | |
| | | % | 16.4 | 16.4 | 5.07 | 49.82 | 3.85 | | | | | |
| | 20-25 | g | 10.3 | 0 | 16.73 | 379 | 49.35 | 423.88 | 477.14 | -12.57 | | |
| | | % | 2.16 | 0 | 3.51 | 79.43 | 10.34 | | | | | |
| | 25-30 | g | 0 | 0 | 11.58 | 423.65 | 55.8 | 460.54 | 502.22 | -9.05 | | |
| | | % | 0 | 0 | 2.31 | 84.36 | 11.11 | | | | 3321.48 | |
| Core 2 | 0-5 | g | 324.65 | 56.13 | 3.01 | 17.79 | 0.37 | 396.97 | 396.98 | -0.003 | | |
| | | % | 81.78 | 14.14 | 0.76 | 4.48 | 0.09 | | | | | |
| | 5-10 | g | 326.9 | 101.43 | 8.99 | 83.34 | 2.4 | 540.38 | 538.84 | 0.29 | | |
| | | % | 60.67 | 18.82 | 1.67 | 15.47 | 0.45 | | | | | |
| | 10-15 | g | 226.4 | 80.76 | 17.64 | 209.14 | 10.34 | 602.94 | 613.25 | -1.71 | | |
| | | % | 36.92 | 13.17 | 2.88 | 34.1 | 1.69 | | | | | |
| | 15-20 | g | 60.33 | 60.33 | 6.5 | 109.45 | 12.3 | 305.29 | 283.15 | 7.25 | | |
| | | % | 21.31 | 21.31 | 2.3 | 38.66 | 4.34 | | | | | |
| | 20-25 | g | 49.16 | 49.16 | 11.75 | 264.67 | 64.89 | 450.22 | 444.4 | 1.29 | | |
| | | % | 11.06 | 11.06 | 2.64 | 59.56 | 14.6 | | | | | |
| | 25-30 | g | 40.7 | 40.7 | 12.91 | 339.78 | 102.32 | 526.5 | 499.2 | 5.19 | | |
| | | % | 8.15 | 8.15 | 2.59 | 68.06 | 20.5 | | | | 2822.3 | |
| Core 3 | 0-5 | g | 309.67 | 219.94 | 2.09 | 9.63 | 0.13 | 360.44 | 360.84 | -0.11 | | |
| | | % | 85.82 | 60.95 | 0.58 | 2.67 | 0.04 | | | | | |
| | 5-10 | g | 276.47 | 76.04 | 14.81 | 96.49 | 3.11 | 536.99 | 538.75 | -0.33 | | |
| | | % | 51.32 | 14.11 | 2.75 | 17.91 | 0.58 | | | | | |
| | 10-15 | g | 4.41 | 4.41 | 17.29 | 113.24 | 9.51 | 286.76 | 288.33 | -0.55 | | |
| | | % | 1.53 | 1.53 | 6 | 39.28 | 3.3 | | | | | |
| | 15-20 | g | 12.15 | 12.15 | 24.09 | 434.37 | 78.81 | 577.59 | 621.58 | -7.62 | | |
| | | % | 1.95 | 1.95 | 3.88 | 69.88 | 12.68 | | | | | |
| | 20-25 | g | 51.94 | 51.94 | 14.85 | 343.34 | 73.53 | 487.19 | 493.01 | -1.2 | | |
| | | % | 10.54 | 10.54 | 3.01 | 69.64 | 14.92 | | | | | |
| | 25-30 | g | 42.33 | 42.33 | 11.39 | 337.06 | 63.9 | 490.96 | 477.93 | 2.65 | | |
| | | % | 8.86 | 8.86 | 2.38 | 70.53 | 13.37 | | | | 2739.93 | 8883.71 |